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Conference on Adaptive Ecosystem Restoration and Management:

Restoration of Cordilleran Conifer Landscapes of North America

June 6-8, 1995

Flagstaff, Arizona

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Abstract

The purpose of the conference was to facilitate the development of mutually beneficial human : wildland interactions by exploring ways in which to restore and sustain land health, as well as that of dependent human communities, in an adaptive ecosystem management context.

Although general adaptive ecosystem restoration and management principles were discussed, the conference was specifically designed to encourage cooperative North American work and to aid in the development of mutualistic interactions between land managers, researchers, administrators, and other individuals and organizations concerned with ecosystem restoration.

The primary focus was on long-needled pine (principally ponderosa and closely related pines) and mixed-conifer landscape systems of the Cordilleran region of North America (the region of North America extending from the Sierra Madre Occidental of Mexico through the western United States to southern British Columbia).

Keywords: ecosystem health, ecosystem management, ecosystem restoration, ponderosa pine, prescribed fire, restoration ecology, wildfire

Compiler's Note: In order to deliver symposium proceedings to users as quickly as possible, many manuscripts did not receive conventional editorial processing. Views expressed in each paper are those of the author and not necessarily those of the sponsoring organizations or the USDA Forest Service. Trade names are used for the information and convenience of the reader and do not imply endorsement or preferential treatment by the sponsoring organizations or the USDA Forest Service.

Cover photos: Forest change at Lick Creek study area, Bitterroot National Forest, Montana, 1909-79. Photo dates are located on the reference tree in each photo. (Complete photo series published in "*Seventy Years of Vegetative Change in a Managed Ponderosa Pine Forest in Western Montana--Implications for Resource Management*," by G.E. Gruell, Wyman C. Schmidt, Stephen F. Arno, and William J. Reich, General Technical Report INT-130, 1982.)
1909: Original stand appears to have been quite open before 1907-11 timber harvest. Analysis of stump at the feet of Ranger Tanner shows evidence of 5 different wildfires prior to logging. Photo by W.J. Lubkin (USDA FS Photo 86475).
1937: View is completed screened by heavily stocked young pines. Pine at right center has died. Photo by K.D. Swan (USDA FS Photo 354396).
1968: Selective cuts and precommercial thinning have occurred, and remaining trees have gained good growth. Photo by Wyman Schmidt (USDA FS Photo 518776).
1979: Bitterbrush and willows have become established; growth of young pines has accelerated. Photo by William Reich.

USDA Forest Service
General Technical Report RM-GTR-278

June 1996

Conference on Adaptive Ecosystem Restoration and Management: Restoration of Cordilleran Conifer Landscapes of North America

June 6-8, 1996

Flagstaff, Arizona

Wallace Covington and Pamela K. Wagner, Technical Coordinators

Northern Arizona University, School of Forestry

Sponsoring Organizations

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Forest Fire Laboratory
Rocky Mountain Forest and Range Experiment Station
U.S. Fish and Wildlife Service
USDA Forest Service - W.O., Fire Management

PREFACE

This conference was designed to explore ways in which to restore and sustain land health, as well as that of dependent human communities, in an adaptive ecosystem management context. The conference was specifically designed to encourage cooperative North American work and to aid in the development of mutualistic interactions between land managers, researchers, administrators, and other individuals and organizations concerned with ecosystem restoration. The conference facilitated the exploration and development of ecological restoration science, practice, and philosophy.

Because the needs and opportunities are so great in the region, the primary focus was on long-needled pine (principally ponderosa and closely related pines) and mixed conifer landscape ecosystems of the Cordilleran region of North America (the region of North America extending from the Sierra Madre Occidental of Mexico through the western United States to southern British Columbia).

An integrating theme was Aldo Leopold's definition of land health restoration as, "a protest against destructive land use"; as "a positive exercise of both skill and insight, not merely a negative exercise of abstinence and caution"; and, finally, as "a universal symbiosis with land, economic, public, and private". Secretary of Interior Bruce Babbitt delivered the keynote address emphasizing the importance of moving ahead with scientifically credible ecosystem restoration as a means of enhancing wildland values.

Conference sessions consisted of:

Day 1: Evolutionary, Cultural, and Scientific Foundations, oral presentations and an evening poster session

Day 2: Ecosystem Restoration and Management Field Trips

Day 3: Putting Adaptive Ecosystem Restoration into Practice, oral presentations

The topics discussed covered the breadth of ecosystem science and management disciplines and involved culturally diverse views ranging from indigenous cultural perspectives to federal, state, and university views as articulated by practitioners and researchers from Canada, the United States, and Mexico. Of particular value was the enthusiastic discussion of approaches to restoration which occurred during the field trips and in informal evening sessions. Written and oral evaluations of the conference indicated that it was highly successful in achieving its goals.

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Opening remarks

M. Jean Hassell[1]

I am honored and pleased to welcome you to this important conference on Ecosystem Restoration. This topic which is near and dear to my heart, deals with a subject and a part of the country to which I have dedicated 42 years of my life.

Before getting into my remarks, I extend a greeting and a welcome to you from Governor Symington. The Governor has taken great interest in the state's natural resources. He understands the importance of these resources to the citizens of Arizona whose daily lives are impacted in one way or another by the way these lands and resources are managed. He is also deeply aware of the national significance of Arizona's varied resources, such as the national parks and monuments, our forests, and our deserts. I hope that my comments today will do justice to the deep sense of responsibility that Governor Symington feels for Arizona's natural resources and the rich cultural mix of Arizona's citizens who depend upon these lands.

While preparing my remarks for this conference, I tried to recall how many times in my career I have participated in other conferences aimed at doing a better job of caring for the land and resources of our citizens. Well, I couldn't remember how many, but I do know that always before, I had a feeling of hope and excitement that the conference I was attending might be the one where all of the competing views, egos, and agendas could be put aside and we could come together and agree on a program of action and direction which would preserve our resources for their own sake and for the sake of our dependent citizens, which includes each and every one of us. This time I don't have the feeling of high expectation, as much as I do a feeling of frustration and frankly some panic. Time is running out as we teeter precariously on the edge of dramatic and sudden change, insofar as our forests and rangelands are concerned. The uses we make of them, economically, socially, and culturally are being questioned as never before.

This conference takes place in a time when much of what I have feared is coming true. Many knowledgeable people, both scientists and experienced laymen, have been predicting these changes now for several years. Let me list for you some of these thoughts:

- There is throughout the West a deep concern for small dependent communities of rural people who depend upon the land and resources of our public and private forest and rangelands.
- Environmental activism is alive and well, and is too often seen as not being in the best interests of the resources or the needs of the people, particularly in rural areas and communities.
- Verbal and legal clashes between the extreme views about how to or whether to manage is all too common today. It is almost a way of life and not a pleasant or productive enterprise in my view.
- This past year the nation spent almost a billion dollars fighting wildfires.
- This past year 26 firefighters were killed fighting wildland fires.
- In 1990, here in Arizona, six firefighters from a state crew were killed on the Dude Fire which destroyed 60 homes, burned 28,000 acres, and cost nearly 12 million dollars to suppress.
- Damage to the soil in the Dude Fire area continues unabated. Erosion will continue for years to come.
- As in the forests and rangelands in much of our nation, fuels in Arizona's forest continue to build. In one of the closely studied and monitored areas just south of Flagstaff on the Bar M Watershed, fuel loading has increased from about 1 ton per acre in 1867 to over 20 tons per acre today!
- Basal areas of live trees have increased from 17 square feet to 154 square feet to the acre.
- There are now some 850 trees per acre on the Bar M, where in 1867, there were on average only 23 trees per acre.
- Herbage growth has decreased on the Bar M from 1,134 lbs per acre in 1867 to an average of 112 lbs per acre now.
- Tree crowns now occupy 60% of the canopy area where they only occupied about 10% in 1867.
- Wildlife habitat and watershed conditions have changed drastically since 1867 and , whether you believe this is good or bad is really not material. The reality is, what we have today is not sustainable!

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[1] *Arizona State Forester, Arizona State Land Department, Phoenix, AZ*

There is a widespread belief among respected scientists and practitioners that a very short period of time remains to change our management of the ponderosa pine system. The current system may be too far removed from its normal evolutionary course to restore its structure, composition, and function. This means these forests, as we see them today, are not sustainable in their present condition. Prior to European settlement these same forests were in balance and had been for hundreds, if not thousands, of years.

As I listen to the various interests, there is little disagreement on the state of forest conditions and what we are facing immediately ahead. In Volume 1, no. 8, 1994 edition of Wild Forest Review, Jeffrey St. Clair's writing about the forests in the southwest says it this way, and I quote:

> "The ecosystem has been terra formed, inverted almost. But it is aiming to right itself of its own volition. The fuel is building up. The fires will come. There will be no stopping them."

I cannot disagree with Mr. St. Clair's statement of conditions in our forests, but he goes on to say,

> "Perhaps, then, the once and future forest will rise up out of the ashes like a true Phoenix. Burn, baby, burn."

My disagreement with Mr. St. Clair is not over the dreadful conditions of our forests, but with his notion that after the fires, we will once again have a beautiful forest. His view glosses over the threat of damage to something isn't done to change our present course. His view ignores the fact that fires of the intensity possible with present conditions will do untold damage to fragile southwestern soils. Watersheds will be eroded and stripped of soil and vegetation. What we now know as a ponderosa pine forest could well be started down an entirely new evolutionary path both unknown and difficult to predict at this time.

Much has been said about how we got into this deplorable situation. Environmentalists blame the Forest Service, ranchers, and loggers; and the forest users blame the Forest Service, environmentalists, politicians, and others for the conditions. The general public is concerned, but really doesn't know quite what is wrong or who to blame. Politicians blame whomever is handy and fits the situation.

I wish I had a small fraction of the time and money that has been spent on finger pointing and blaming the next guy, so we could reapply it to finding solutions. I believe the truth of the matter is, people did the best they knew how to do at the time. I hope we have learned something in the process and can now stop the finger-pointing and blaming so we can get started with some positive unified action.

Knowledgeable people agree there is a problem. We need to work together for a middle-ground answer. The 'burn, baby, burn' prescription is not going to be acceptable to the public at large. The past methodical prescriptive management has not worked, we need something different. I don't know the best name for what we need as a guiding light. Ecosystem management is broad and somewhat ethereal and hard to measure, but if it does mean dealing with all of a system's components on a sustainable basis and not just a single function in the short run, then I am all for it. I like the notion of adaptive restoration of ecosystems, which to me means starting with a definable base and vision for what is desired and learning from our actions as we go along. It is the opposite of prescriptive rigid management. We monitor the effects of our actions and adapt to what we learn.

There is a place and a need for all views and ideas if we are to use time and resources in a productive way. Environmentalists are not always wrong, loggers and ranchers are not always right. We all have a stake in the outcome.

My plea to the participants in this conference is this:

- Let's stop the feuding.
- Let's do something together that will restore health to all parts of our ecosystem.
- Let's not burn it down with the false expectation everything will just be all right.
- The public cannot, will not, and should not live with that direction.

On behalf of Governor Symington and myself, this conference holds out a hope that we can and will come together in setting a direction for our forests and rangelands that will preserve our soil, flora, and fauna, which in turn will continue to sustain the citizens of our culturally rich state. I know there are some of you on either side of the issue who feel you just cannot bear the notion of compromise and working together; but if we don't pull together, we will all lose in the long run. Let's work together to find an amicable, lasting solution to the problems we face. Time is of the essence!

ECOSYSTEM DISTRESS SYNDROME IN PONDEROSA PINE FORESTS

David J. Rapport[1] and Sergei B. Yazvenko[1]

ABSTRACT. Ponderosa pine forests in the western U.S. were maintained open and park-like by frequent low-intensity fires and pest outbreaks. Fires promoted dense grass cover, mineralized organic matter, and eliminated weak trees. European settlers changed the balance by practicing heavy grazing which damaged grass cover and triggered soil erosion and depletion of nutrient pool. Later, fire suppression promoted establishment of dense pine thickets and excessive fuel accumulation. Today, many forests exhibit Ecosystem Distress Syndrome (EDS): slow production and growth, decreased nutrient cycling, declining biodiversity, simplified structure, increased soil erosion, and increased rates of diseases. Many ecosystem services and management options are lost. Restoration efforts must aim to eliminate EDS, but the truth is that even relatively simple and disturbance-dependent ecosystems are difficult and costly to repair once the EDS becomes pronounced.

WHAT IS FOREST ECOSYSTEM HEALTH?

Ecosystem health has been a recognized topic for discussion for some time, yet, the notion remains controversial (Steedman 1994; Kimmins 1995). There are three widespread if not well defined concepts of forest health: "utilitarian", "ecosystemic" (Kolb et al. 1994), and "historical" (Kimmins 1995). The utilitarian view focuses on timber production, and recognizes a healthy forest if trees are vigorous and productive, biomass accumulation and nutrient cycling are rapid, pests are suppressed and unimportant, and "vigor" or "vitality" is high. This approach may be relevant when management objectives are clearly defined, e.g., in industrial forests. It is explicitly subjective, and the judgment depends on management goals.

The ecosystem approach focuses on the ecological processes and structures that maintain forest condition (Kolb et al. 1994): biodiversity and ecodiversity; physical environment, biotic resources to support long-term productivity; resistance to and the ability to recover from impacts at the landscape scale; a functional equilibrium between supply and demand of essential resources. Other definitions stress the importance of history (see Kimmins 1995): the forest is looked upon as healthy if the ecosystem structure and processes at the stand level are within the historical range of variation characteristic for that ecosystem. This approach underscores the vital necessity of very long term studies of forest ecosystems.

In the present paper, we are defining our understanding of ecosystem health and examine a specific case of ponderosa pine forest ecosystem in the southwestern U.S. In the context of this study, we explore the mechanisms by which European settlement altered ponderosa pine forests from their natural condition to the present state. Three complementary approaches will be utilized (Rapport 1989). One involves examining the signs of Ecosystem Distress, including decreased nutrient capital and rates of nutrient cycling, decreased biodiversity, increased soil erosion and rates of diseases, etc. (Rapport et al. 1985).

We then examine the resilience of ecosystems, or their capacity to rebound from normal disturbances. We view resilience as an indication of ecosystem health, and the data suggest that currently, the resilience of ponderosa pine forests is seriously compromised. The third approach involves risk analysis with the focus on stress factors rather than on ecosystem responses. We conclude that the health of ponderosa pine forests has dramatically deteriorated since Euro-American settlement, and prolonged and costly efforts are needed to restore its pre-settlement condition.

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Ponderosa pine forest ecosystem today

Commercial forest dominated by ponderosa pine (*Pinus ponderosa* Lawr.) occupies almost 24 million acres in the western U.S. (Blake et al. 1989). A typical unlogged ponderosa pine forest in Arizona today is a mosaic of stands of old growth trees divided by dense stagnant thickets of small trees of ponderosa pine. Thickets are uncommon below old trees, perhaps due to the high accumulation of litter and duff. In the Southwest, many of the thickets originated around 1919, when a rare combination of factors occurred (Schubert 1974). A good seed production in 1918 was followed by unusually moist springs for a few years which benefited the establishment and growth of pine seedlings. Moreover, survival rate of the seedlings was enhanced by the previous heavy grazing which damaged grass cover, and suppression of fires which would have eliminated most young trees (Madany and West 1983; White 1985). The successful natural establishment of ponderosa pine in the Southwest has always been a rare event. The factors necessary for this process are infrequent by themselves, let alone their proper combination and timing. Natural controls include irregular seed production, high rate of cone damage from insects (Blake et al. 1989), frequent spring drought damaging seedlings, suppression by perennial bunch-grasses, and site-specific factors.

Soil moisture is perhaps the critical factor in establishment of ponderosa pine in the Southwest. Grasses deplete soil moisture faster than seedlings of ponderosa pine, thus retarding the growth of the seedlings. Allelopathic relations have not been proved by direct evidence but can not be ruled out either (Schubert 1974; Madany and West 1983).

Recent dynamics of the ponderosa pine forests

Presettlement condition

It was believed that prior to European settlement, ponderosa pine in the Southwest grew in even-aged groups established following a disturbance (fire or pest outbreak) which replaced an entire previous group (Cooper 1960). White (1985) studied a 7.3-ha old-growth stand of ponderosa pine in northern Arizona and found out that it was not the case in the area. White obtained tree-ring data on all ponderosa pine trees established prior to European settlement (before the 1870's). No uniform age groups of trees were found, and age variation within a group was 33-268 years. These results suggest that fires in the area were of relatively low intensity and were usually unable to eliminate the whole stand. Pine seedlings became established when 1-2 large trees in a group died, fell, and provided enough fuel for an intensive burn of the spot. Hot fire suppressed grasses for the time sufficient for the establishment of pine seedlings, provided other conditions (mainly good seed production and moist spring) were met. The fact that only 22% of the total study area was occupied by old-growth pine trees supports the idea that only restricted "safe areas" (Harper 1977) were suitable for tree regeneration. Today, old snags and logs are mostly confined to the areas occupied by trees today (White 1985), and there is little evidence that groups of trees ever grew outside their present sites.

It can be stated that before European settlement, fire was one of the major factors controlling and maintaining ponderosa pine forests. Fires in pre-settlement ponderosa pine forests in Arizona were frequent (Dietrich 1980) and were the principal cause of the park-like appearance of the forests (Cooper 1960) by burning pine litter and duff, promoting grass growth, and altering various ecosystem parameters:

- Species composition (Oswald and Covington, 1984; Vose and White 1987),
- Soil nutrients (Ryan and Covington 1986),
- Soil moisture (Haase 1986),
- Forest floor volume (Covington and Sackett 1984),
- Understory productivity (Gaines et al. 1958; Vose and White 1991).

Fire may also influence the phenology of understory, because it increases soil surface temperature by removal or thinning of the litter layer, darkening floor surface and/or opening the canopy (Daubenmire 1968).

European settlement

When ranchers settled in northern Arizona in 1870s, they changed the natural balance of ecological forces (Cooper 1960; White 1985). Combinations of such factors as overgrazing, fire suppression, and unusually moist years in the 1920's contributed to the conversion of the landscape from savanna or patchy forest into an almost continuous forest cover. Today, forest often accumulates large fuel loads, has scattered grass cover (White 1985, White et al. 1991), reduced herbal production and diversity, and increased soil erosion (Weaver 1951; Savage 1991; Covington and Moore 1994). In many areas ponderosa pine forests are undergoing a shift of dominance towards slowly growing shade toler-ant species, which may represent later succession stages (Weaver 1974). Swezy and Agee (1991) report that in central Oregon, after about 80 years of fire suppression, white fir (*Abies concolor* (Gord. and Glend.) Lindl.) dominates the stands where originally mixed open forest of old-growth ponderosa pine and white fir existed.

Forest dynamics under various fire regimes has been simulated for the forests dominated by ponderosa pine and Douglas fir (*Pseudotsuga menziesii* (Mirbel) Franco) forests in the Inland West (Keane et al. 1990). The results suggest that under frequent fires ponderosa pine and western larch (*Larix occidentalis* Nutt) become dominant species. If fires are suppressed completely, Douglas fir is gradually established in the overstory and inevitably becomes the dominant species. This shift is facilitated by the selective harvesting of large pines which has been the case in much of the area. Natural fire regime with occasional long inter-fire intervals maintains a proportion of Douglas-fir in the overstory, while pine and larch remain dominant species.

Summarizing, it seems likely that changes brought about by human impact on ponderosa pine forests are intricate combinations of influences, some of which lead to the establishment of later successional stages, while others indicate that the ecosystem is stressed. Ponderosa pine forest ecosystem have been released from a major natural disturbance (fire) while it is subject to anthropogenic stresses (logging, overgrazing, road construction, etc.). Thus, the symptoms of changes are not simple and will be analyzed in some detail. As for any forest ecosystem, only long-term studies can provide a time perspective sufficient for distinguishing between normal fluctuations and those outside the historical range of variation. Because an old-growth forest is dominated by trees centuries old, tree-ring analysis becomes the key method available to assess the full range of multicentury historical variation.

Ecosystem health approach to assess ponderosa pine forest health

To assess forest ecosystem health, we suggest three complementary approaches (Rapport et al. 1985). The first approach involves screening for signs of Ecosystem Distress. Usually, Ecosystem Distress Syndrome (EDS) becomes pronounced in heavily damaged ecosystems, when restoration costs are often prohibitive (Maini 1993). The second approach attempts to assess counteractive capacity of an ecosystem (resilience or stability). This is an ecosystem analog of a fitness test in medicine, capable of gauging ecosystem health prior to the onset of its breakdown under excessive stress. The third approach involves risk analysis and focuses on monitoring stress factors rather than ecosystem responses. With some attendant uncertainty, it allows an assessment of forest health "before the fact".

General Screening Indicators: Ecosystem Distress Syndrome (EDS)

Health is assessed by the presence or absence of indicators, i.e., specific parameters of an ecosystem which are believed to reflect key aspects and processes in the ecosystem. Since a single indicator alone is generally unreliable (whether in human or ecosystem health assessment), a suite of indicators is generally assessed. An ecosystem may be viewed as healthy in the absence of signs of EDS (Rapport 1989; Hilden and Rapport 1994; Costanza 1992). Table 1 demonstrates the presence of many signs of EDS even in unlogged ponderosa pine forests.

Primary production/growth. Assessing long-term growth dynamics at the landscape to regional scales is quite difficult, and the results of such studies are not unambiguous. Peterson et al. (1991) studied regional growth changes in ozone-stressed ponderosa pine in the Sierra Nevada. Century-scale growth patterns were evaluated using tree-ring analysis, in order to provide the temporal context to possible recent changes. The study's main question was whether there were growth changes since the 1950's (when substantial air pollution

Table 1. Presence of signs of Ecosystem Distress in ponderosa pine forests.

Indicator	Behavior	Reference
Primary production/tree growth	decreases	Oswald and Covington, 1984; Peterson *et al.*, 1991; Temple and Miller, 1994
Rates of decomposition	decreases	Covington and Sackett, 1992
Rates of nutrient cycling	decreases	Covington and Moore, 1994
Soil erosion	increases	Savage, 1991; Covington and Moore, 1994
Rate of diseases	increases	Fenn *et al.*, 1990
Amplitude of fluctuations	increases?	
Biodiversity:		
Species richness	decreases	Uresk and Severson, 1989; West, 1993
Species functional dominance	shifts	Cooper, 1960; Dickman, 1978; Madany and West, 1983; White, 1985
Regression to opportunistic species	occurs?	Madany and West, 1983; White, 1985
Reduction in size	absent	White, 1985; Covington and Moore, 1994

started) outside the range of natural variation in growth. Broad regional scale correlation between ozone concentration and foliar injuries in ponderosa pine has been demonstrated. However, no evidence of recent regional scale growth changes exceeding the historical range of variation was found. This result contrasts with the findings of recent growth increases for subalpine tree species in Sierra Nevada (Peterson et al. 1989). A substantial number of ponderosa pine trees actually decreased in growth in the 1950's and 1960's, but in an even larger number of trees the growth increased.

So the hypothesis of the large scale influence of air deposition on growth of mature trees was not substantiated in this case. However, premature needle senescence in heavily affected areas, and hence, decreased productivity, was well documented. This study reveals one of the few examples of widespread injury associated with non-point source of air pollution in the North America. The only other example is the regional scale damage of fir and pine forest southwest of Mexico City (Cibrial Tovar 1989). However, the challenge of distinguishing the influence of ozone or other air pollutant from the effects of other factors (droughts, stand development, silvicultural practices such as thinning) was not met within the framework of the study.

Rates of decomposition and nutrient cycling. In many cases, slowing rates of decomposition serve as an early warning sign of pathology in forest ecosystems (Bormann and Likens 1979). In ponderosa pine forests, the current accumulation of organic matter (litter, duff, and coarse organic mater) indicates retarded decomposition and cycling, because the nutrients are locked in the form unavailable for utilization by plants. Pine litter and wood is rich in lignin, a generic inhibitor of microbial activity. Fire accelerates nutrient cycling mainly by mineralizing nutrients, and fire exclusion inhibits this process.

Rate of diseases. Various pests have been interacting with ponderosa pine forest ecosystems for millennia, having formed a stable prey-predator type system. Long and Wagner (1992) studied the impacts of southwestern pine tip moth *Rhyacionia neomexicana* (Dryar) on ponderosa pine. Tip moth infests and damages terminal and lateral shoots of young trees and causes considerable losses in tree growth (Jennings 1975). Comandra blister rust is another widespread pest infesting many tree species, lodgepole pine among them. Significant correlation of disease rate with the tree age has been demonstrated (Jacobi et al. 1993). Unfortunately, these kinds of studies are limited by the absence of long-term records, so the context for the interpretation of the results is lacking. This deficiency is highlighted by the dendroecological studies showing remarkable stability of forest-pest systems. Native pests co-existed, interacted and co-evolved with their hosts, so by today these systems evolved into mature prey-predator cycles which ultimately stabilize and benefit both components.

Whether the most recent outbreaks exceed the historical range of ecosystem fluctuations, was one of the goals of a study conveyed in northern New Mexico (Swetnam and Lynch 1993). The authors investigated the multicentury history of western spruce budworm (*Choristoneura occidentalis*) using tree-ring sequences from 24 mixed conifer stands over the region. The primary host is Douglas fir, but the results may also be relevant for other forest types. The regional scale design of the study was insightful in several aspects. The authors were able to detect local to regional scale patterns in forest disturbances. The forest-budworm system dynamics appeared to be pseudoperiodic at the regional scale with the return period 20-33 years. The duration of infestations within stands was about 11 years. One ancient stand more than 700 years old was shown to have sustained at least 20 outbreaks and yet the forest-budworm system was apparently stable.

The long-term impact of the budworm has been significant. It selectively infests spruces, thus favoring pines. Perhaps overall forest density decreases while nutrient cycling is accelerated because of the mass of frass and dead bodies falling from trees, and increased light availability favors surviving trees and understory species. So infestation may possibly be of importance comparable to that of fires (Swetnam and Lynch 1993). The most recent outbreak of the budworm which began in the 1970's has been so severe, it probably exceeds the historical range of variation (Swetnam and Lynch 1993).

Resilience (counteractive capacity) of ponderosa pine forest ecosystems

Resilience can be termed as the "return time" (Rapport 1995) of an ecosystem rebound after disturbance. It is assumed that the healthier the ecosystem, the greater its capabilities for recovery and the faster it rebounds (Rapport 1992; Rapport 1995). One test of this hypothesis is to study an area which was harvested and also subjected to other stresses, and compare its rebound with a similar area which was harvested but not subjected to other stresses. This "fitness test" ought to be feasible providing that long-term data are available and the parameters of rebound are selected carefully. Recently, such tests were carried out for the grasslands of New Mexico and their results supported the hypothesis (Rapport et al. 1995).

The natural resilience of ponderosa pine forests and other conifer dominated forests in the Southwest was high due to the fact that historically, they were subject to periodical disturbances: fires or pest outbreaks. Moreover, the forests actually depended on the disturbances in order to maintain their integrity. Johnson and Fryer (1989) have studied the long-term population dynamics

in montane lodgepole pine (*Pinus contorta* Dougl. ex Loud.) and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) stands using dendroecological techniques. They have found that the populations of both species are not self-reproducing, because the recruitment does not compensate for the mortality. However, the populations are not transient either: the fire frequency is higher that the life-span of the fire-cohort, so, the long-term dynamics of both species is controlled by the fire frequency and the rate of establishment of fire-cohorts.

The study of the fire history in a ponderosa pine/ Douglas fir forest in Colorado (Goldblum and Veblen 1992) revealed that mean fire intervals were 31.8 years for the pre-settlement era (before 1859), 8.1 years for the settlement era (1859-1920) and 28 years for the fire suppression era (after 1920). The increase in frequency of fires in the early settlement time has been attibuted to anthropogenic fires. However, the pre-settlement fire frequency was probably underestimated, because few trees from that time survived either alive or as logs.

The mounting evidence on the critical role played by the natural fire regime in maintaining healthy ponderosa pine forests has led researchers and forest managers to use prescribed fires as a recipe to improve the current condition of ponderosa pine forests. The outcome of early experimentation has been the realization that post-fire mortality among pine trees is often prohibitively high (Swezy and Agee 1991). The mortality has been found to be affected by bole charring (Ryan and Losensky 1988), tree size (Van Wagner 1973), the season of fire, postfire weather, and insect attack.

The accumulation of duff because of long fire exclusion is important in considering the consequences of prescribed fires. In some conifer forest types of the western U.S., as much as 60-80% of the biomass of fine roots occurs in the forest floor. In a ponderosa pine forest in central Washington, a burned plot demonstrated more than 60% decrease in fine root biomass during a two-week period since a prescribed fire, while unburned control area increased its fine-root biomass twofold. This difference is likely to contribute later on to a decreased biomass accumulation and even dieback of mature trees on the burned plot. A similar pattern of post-fire tree mortality has been observed in an unlogged ponderosa pine forest (Keen 1936): among the young and immature age classes, low-vigor trees had the highest mortality rate, while among older trees only the most vigorous ones survived. A high mortality (over 30%) was possibly caused by the fact that the ecosystem may not be adapted to the spring fires, as natural lightning fires occur mostly between July and September. Also, the present fuel loads seem too hazardous to secure desired mortality among white fir while maintaining relatively low mortality of old ponderosa pine trees (Swezy and Agee 1991).

The resilience of the ecosystem seems to have been seriously compromised since Euro-American settlement, so that fire, once a regular disturbance, has become a damaging distress. The main components of this transformation are probably high fuel loads and decreased vitality of individual trees. In central Oregon, after about 80 years of fire suppression, white fir (*Abies concolor* (Gord. and Glend.) Lindl.) dominates the stands where open forests of old-growth ponderosa pine and white fir existed before (Swezy and Agee 1991). Prescribed fires conveyed in spring and autumn cause excessive mortality among larger trees of ponderosa pine, and the effects become obvious only several years after the fire, with bark beetles being a major direct cause of mortality. Because of the loss of old-growth ponderosa pines, the dominance has shifted towards white fir, thus resulting in the situation the opposite to the goals of the prescribed fire program (Swezy and Agee 1991).

Implications to rehabilitation of forest health

Implications of the framework sketched above for the rehabilitation of the western forests are numerous.

Firstly, once an ecosystem has been so damaged as to lose its resilience and display many of the signs of EDS, it is likely too late to rehabilitate it at any reasonable cost (Maini 1993; Rapport et al. 1995).

Secondly, in monitoring the success of rehabilitation, explicit criteria for forest ecosystem health should be designed, in order to be able to answer the following questions: Do specific signs of EDS disappear? For example, if one sign of damaged forest ecosystems is increased disease prevalence -- does this decline significantly to base-line levels? Has the tree species diversity and age structure been altered to a more sustainable (and desirable pattern)? Has soil fertility improved? Such criteria can be objectively measured and used as a basis to gauge rehabilitation.

Ecosystem health, however, is more than the bio-integrity of the system. It also involves economic health and human health. Thus questions in these aspects can also be formulated to gauge rehabilitation success. For example, have risks—perhaps owing to the use of pesticides to control infestations—been reduced? Has the quality of drinking water been improved? Has economic and cultural opportunity been enhanced?

A further check on the restoration of forests would be an enhancement in the capability of forests to fully recover after disturbances, or the "fitness" of forests (Rapport et al. 1995). In some forest types there is evidence that severe harvesting and other sources of disturbance have resulted in the loss of resilience (Bird and Rapport 1986).

Conclusions

1. Assessments of forest ecosystem health depend first on the development of the concept and definition of health, and its applications to forest systems. There are a variety of approaches and all of them have their merits and limitations.
2. The concept of ecosystem health has other vital dimensions in addition to the bio-physical one: societal, economical, cultural, spiritual, etc.
3. The approaches to preserve and restore forest health have to be concrete, depending on a forest type and the temporal/spatial scale under consideration. This necessitates the use of regional analysis that takes into consideration the specific combination of ecological factors in every area.
4. One of the central guidelines for maintaining forest health is that the management mimic the pattern and scale of disturbances to which the ecosystem is historically adapted.
5. The restoration criteria can be established based on ecosystem health criteria. The appearance of the EDS, the loss of resilience, and risk factors, are important considerations in setting the objectives for a successful restoration program.

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Nouvelle southwest

Stephen J. Pyne[1]

Abstract. The new Southwest is a product of the old. The region boasts an ideal formula for natural fires. Its dramatic terrain and well-defined wet-dry cycles, both annual and secular, have long established it as an epicenter for lightning fires. But the real narrative of Southwest fire history belongs to its human firebrands, who have co-existed, if not co-evolved, with the regional biota throughout the Holocene. Different waves of human colonization have shaped distinctive fire regimes.

The Old Southwest simmered over chronic fires like a cauldron. That pattern changed abruptly with European-derived settlement. Grazing provided the primary shock wave; but high-grade logging, fixed land ownership, connections to national markets, the reservation of lands for parks, forests, and native tribes, the introduction of exotic flora, and urbanization, all redefined the relationship of humans to the lands around them and in so doing have altered, by means both direct and indirect, the regional fire regimes. Increasingly the landscape has assumed forms that humans find unattractive, even threatening. Increasingly the region's fires appear more unmanageable. Overall, burning has decreased, and where it survives, it displays a severity not previously experienced and an ecological synergy very different from prehistoric conditions.

But however significant fire was to the Old Southwest, its removal was only one part of fashioning the New Southwest, and reinstating fire will not by itself restore that old landscape. It is not obvious by what means fire should be reinstated, nor to what ends it should be applied. For better or worse the standards for fire regimes reside in contemporary humans, not in nature and not in history.[2]

The fire bust of June 1990 followed a classic formula but one intensified in puzzling ways.

In the Southwest fire seasons follow a natural rhythm of wet and dry, a two-cycle engine for which lightning typically provides the spark. This interplay of wet and dry takes several forms. Part is topographic, in which the Southwest's fabulous terrain creates differentials between moisture and aridity. Slopes that face south or north betray different levels of moisture. So do plateaus that range from high to low. So does the contrast between peak and ravine. Someplace is nearly always dry and nearly always wet. The greater cause, however, is climatic, the summer monsoon. Surges of moisture from the south strike mesa and mountain, and thunderstorms tower up like spumes of surf on a reef. Rain descends in drying veils. Lightning kindles fires that flash from the peaks like beacons.

The process is spotty, like a handful of popcorn scattered on a skillet. One moment there is a deluge, the next a flood of desert sun. Ideally, there is enough storm to hurl lightning and wind, not enough to quench burning snags. Add to this winds that splash out from thunderheads like water from an overturned bucket, spillage that makes for dust storms in the desert, fire storms in the mountains. Altogether it is one of the great ecological rituals of the region, and it accounts for the fact that the Southwest has the highest concentration of lightning fires in the United States.

The figures are astonishing. Between 1960 and 1974 there were 12 days in Arizona and New Mexico when more than 100 lightning fires started; on June 28, 1960 lightning kindled 143 fires. In 1970 lightning ignited 100 fires on July 18, and the next day brought 100 more. On June 24, 1971 103 lightning fires burned 75,713 acres. The Southwest's national forests average more fires per year than any other region; they have the second highest rate of burned acreage, from both wild and controlled

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fires; and critical fire weather occurs here with greater frequency and persistence than anywhere else in the nation. Yet despite prodigious numbers of fires, there are few truly devastating burns. The sheer number of ignitions, plus the exquisite minuet between rain and fire, assures a certain equilibrium that balances large numbers of fires with smaller sizes of individual fires. As the monsoon persists, the wet triumphs over the dry. The number of fires rises steadily from May to August while the average size of those fires diminishes.

But while fire busts are frequent, they are rarely immense and even more rarely fatal. The Dude Creek fire outside Payson, Arizona in late June 1990 shocks because it was both. It took familiar elements—some ancient, some modern—and compounded them into an event that looked grotesque, alien, out of character. Perhaps it was. The conflagration savaged forests, houses, and a fire suppression organization.

The fire blew up on the hottest day ever recorded in Phoenix (122.5°), which argued that it was an old story intensified. But it burned through a forest vastly different than in presettlement times, through summer homes and trailer parks that had little historic precedent, through a society that had sought to eliminate fire of all kinds. Old and new had come together with as much force and fury as wet and dry. When a microburst of wind drove the fire through a squad of firefighter inmates, killing six and hospitalizing four others, it prodded debate about how fire and American society could coexist here, which is to say, about what the character of each had become.

That extraordinary fire load is not simply a product of natural processes. For millennia humans have busily restructured the geography and seasonality of Southwestern fire—sometimes complementing and sometimes countering the natural order. Lightning had to compete not only with rain but with aboriginal firesticks. Human inhabitants added other sources of ignition in the service of hunting, raiding, foraging, and horticulture, and as an inadvertent by-product of a seasonal nomadism whose routes became trails of smoke from camp fires, signal fires, and escaped fires of diverse origins.

"The most potent and powerful weapon in the hands of these aborigines," concluded S.J. Holsinger, of the General Land Office, at the turn of the century, "was the firebrand. It was used alike to capture the deer, the elk, and the antelope, and to vanquish the enemy. It cleared the mountain trail and destroyed the cover in which their quarry took refuge." Obviously burning on this scale "must have exerted a marked influence upon the vegetation of the country. Their fires, and those of the historic races, unquestionably account for the open condition of the forest... The high pine forests were their hunting grounds, and the vast areas of foothills and plateau, covered with nut-bearing pines, their harvest fields..." It is important to note that the aboriginal fire regimes were themselves in transition as peoples migrated into or departed out of the region.

There is an old adage in firefighting which says that the fine fuels drive the fire. Fine fuels include grasses, conifer needles, low shrubs, the portion of the fuelbed that reacts most quickly to changes in moisture and heat, that most readily combusts. It determines the ease of ignition and the rapidity of fire spread. Under aboriginal rule, fine fuels blossomed, and the Southwest burned easily and often. Lightning and firestick competed to see which would burn a particular site or in what season. The density of that competition fashioned, like bees in a hive, an intricate honeycomb of burned and unburned sites. In dry years fires simmered for weeks, smoldering and flaring as the opportunity permitted. The principle check against conflagrations was simply the magnitude of low-intensity burning on all sides.

There are eyewitness accounts to the burning, but the most compelling evidence was recorded in the land itself, the golden grasslands, hillside montages of brush and grass, and most spectacularly oak and pine savannas. Early explorers spoke enthusiastically about the great natural parklands of the region in which mature ponderosa pines marched in majestic columns. In 1882 Capt. Clarence Dutton, exploring the Kaibab Plateau for the U.S. Geological Survey, exulted that:

> "The trees are large and noble in aspect and stand widely apart... Instead of dense thickets where we are shut in by impenetrable foliage, we can look far beyond and see the tree trunks vanishing away like an infinite colonnade. The ground is unobstructed and inviting. There is a constant succession of parks and glades ... the pines standing at intervals varying from 50 to 100 feet, and upon a soil that is smooth, firm, and free from undergrowth. All is open, and we may look far into the depths of the forest on either hand."

For his report on a prospective wagon road through northern Arizona, Army surveyor Lt. E.F. Beale wrote in 1858 that:

> We came to a glorious forest of lofty pines, through which we have traveled ten miles. The country was beautifully undulating, and although we usually associate the idea of barrenness with the pine regions, it was not so in this instance; every foot being covered with the finest grass, and beautiful broad grassy vales extending in every direction. The forest was perfectly open and unencumbered with brush wood, so that the traveling was excellent.

The apparent explanation for the character of these semi-tended fields is that only a fraction of ponderosa pine seedlings survived the regular onslaught of fire through bunch grass. Great trees that toppled over ripped up the ground at their roots, creating pockets of grass-free soil for a few years; so did the fallen trunks

when, after a period of decomposition, they burned to white ash. In the critical years that followed, seedlings thrived, and reached a state in which they could survive routine fires. Around Flagstaff, for example, fire-scarred pine testify to fires that burned an average of every 1.5 years. Mature trees grouped oddly, clustered in ways that betrayed their origin in the churned-up soil of old root-holes, or aligned along the trend of fallen boles. The macro-geography of such forests depended on the micro-geography of fire refugia.

The paradox that the land was both burned and forested baffled some observers, like the Norwegian naturalist and explorer Carl Lumholtz who witnessed an astonishing profligacy of aboriginal burning across the border with Mexico. "These Indians, the pagans as well as the Christians, keep up the custom of burning off the grass all over the sierras during the driest season of the year... [so that] fires are seen continually burning day and night all over the mountains up to the highest crests, leaving the stony ground, blackened and barren, but the forests stand green." That was the rub. Despite this fantastic amount of firing, Lumholtz became convinced that "the continuous, immense forests here could never be destroyed by the Indians" because, paradoxically, all this chronic burning inoculated the forests against wildfires. They ensured the forests' "indestructibility."

All this changed with the advent of European colonization. Settlers introduced some new ignitions and removed several old ones, but it was by utterly restructuring the regional fuel complex that they remade the fire regimes of the Southwest. Generally colonization made itself felt in the New World primarily through agriculture. This impact was muted in the Southwest, however. Regional aridity, hostile tribes, distance from major markets, the slow movement of westering Americans, the retarded admission of Arizona and New Mexico into statehood—which meant that the vast majority of lands remained public—all militated in the Southwest against the kind of pervasive agricultural settlement that typified most of the American frontier. Logging and landclearing remained relatively local; farming concentrated on irrigation rather than lands fire-flushed for nutrients. Indians, sequestered onto special reservations, became inconsequential as a source of fire. When Gifford Pinchot visited Arizona in 1900, he watched a distant Apache setting "the woods on fire" to improve his hunting, trailing fire like a broken lance in the dirt. Instead, settlement followed hoof, not axe. Pastoralism prevailed—first through Hispanics, then through partial adoption by select tribes of indigenes (like the Navajo), and then, with mounting force, through Americans. Livestock came to the Southwest in immense numbers. Cattle and sheep—the sheep were "ten times worse," Pinchot insisted—hit the region like a shock wave, dissembling fire regime after fire regime in ways that may prove irreversible. Flocks roamed in the hundreds of thousands, pounding forests and prairies, leaving clouds of biotic dust to blow in their wake. (Pinchot was wrong: the cattle were worse because sheep often browsed on tree seedlings and cattle did not, allowing that arboreal reproduction to blossom unchecked.) When the big ranches collapsed, hundreds of smaller homestead ranches took up the slack. The epidemic of herds continued.

In the old Southwest grass had infiltrated every landscape, and some it dominated. Now exotic herbivores seized every blade and pursued the succulent grass into every niche. No place was spared. Rolling hills of oak, high desert grasslands, mountain meadows, slopes dappled with chaparral, open pine forests like the columns of the great cathedral of Seville—the relentless hoofs and hungry teeth found them all. Thanks to his herds the reach of the rancher far exceeded his own numbers. He became the biotic conquistador of the Southwest. There was no sanctuary, no refugia from the conquest. The indigenous fires went the way of the grizzly bear and the mountain lion. Only on Indian reservations could it survive in anything like its former state and then only if tribes did not take up herding with the same ruthlessness evident elsewhere in the region. What had fed the flames of fast combustion now stoked the slow combustion of metabolizing livestock. Well before systematic fire control cattle and sheep cropped fire from the land, and they did so with a thoroughness that later engine companies, smokejumpers, and helitack crews could never equal.

The evidence is written widely if complexly in the land. The finely bounded mosaic that had constituted the Southwest scene smeared; desert succulents, mesquite, juniper, and chaparral replaced grasses in desert basins and across high plateaus; brush congealed into jungles; open forests, once dappled with glades of sun and shadow, snarled with downed logs, dense tangles of understory, and young groves of pine and fir reproduction "thick as the hair on a dog's back." In 1902 S.J. Holsinger observed that "in Arizona you will find no young forests of any considerable extent antedating a period of forty years, and almost all of the regrowth has sprung up within the last quarter of a century." Surveying southern Arizona in the early 1920s, Aldo Leopold reasoned that "one is forced to the conclusion that there have been no widespread fires during the past 40 years." Forty years later Charles Cooper mapped the peculiar age structure of Arizona pine and determined that the forest derived from a small number of cohorts, all of which became established during the favorable climatic periods of the early 20th century but which survived because they were spared fires. Others have disputed the climatic argument for broadcast regeneration—1919 as an *annus mirabilis*, for example—but

agreed with the outcome. What had been restricted to microsites and select times now thrived everywhere, year after year. Trees and brush multiplied like fruit flies in a jar of bananas. They spread, a scabby reaction to a vast ecological infection.

If it is not everything, timing accounts for much of this condition. An interesting study has compared the forest structure on the Chuska Mountains in the Navajo Reservation to that elsewhere in northern Arizona. The Navajos acquired livestock from the Spaniards, and their herds soon swelled. Huge numbers of sheep and goats came to the Chuskas by the 1820s. As the flocks advanced, fires receded. But the rapid reforestation that followed American grazing later in the century did not immediately occur here. Regeneration apparently had to wait for a favorable climate, which occurred throughout the region in the early decades of the 20th century. The congestion of the Chuska forests thus came synchronously with that elsewhere in the Southwest, the product of a beneficent climate that promoted regeneration and the absence of fire thinning made possible by intensive grazing.

Grazing had plenty of accessories. Loggers aggravated the scene by culling whatever mature or old growth timber they could reach. Bark beetles, fungi, and dwarf mistletoe infested the thickets that, in the absence of grass and fire, sprang up in unnatural profusion. Droughts like that which gripped the region at the turn of the century magnified grazing's shock wave, further reducing the light fuels needed to carry fire. So did the other instruments of settlement, the fire-breaked roads, the patterns of fixed land ownership that prevented the seasonal cycling of peoples and fires, the introduction of exotic flora, the mining and urbanization that created local markets for livestock and the railroads that bound herds to national ones, the reservation of public lands and the establishment of professional forestry. For all their different means, however, they tended to lead to the same end, the suppression of light fire and the encouragement of fuel arrays that promoted intense fires.

The fine fuels—the grasses and forbs—that had carpeted aboriginal Arizona now massed into three-dimensional jungles that readily transformed surface fires into crown fires. In places young growth—two meters high and 70 years old—existed in a comatose state, unable to grow and unwilling to die, waiting until fire could shock them back to life. A century after Beale rejoiced in the open pinelands of the region, Wallace Covington and Margaret Moore estimated that tree density had exploded from 23 per acre to 851, tree basal area from 23 to 315 ft^2 per acre, and crown closure from 8% to 93%. Where Dutton had praised the wooden colonnades of the Kaibab, tree densities had ratcheted upward from 55.9 to 276.3 per acre, tree basal area from 44 to 245 ft^2 per acre, and crown closure from 16% to over 70%. Herbage had correspondingly plummeted, from 1000 to 112 tons/acre in northern Arizona, and from 589 to 117 on the Kaibab. Near Flagstaff a site that herbaceous plants had once covered 83% in 1876 covered only 4% in 1990. More ominously fuel loads rocketed from 2 and 0.2 tons/acre, respectively, to 44 and 28 tons/acre. The land metamorphosed from a pine savanna to a forest tangled in dog-hair thickets. In recent years the woody invasion has also included houses. While the ultimate reasons for this biotic drama reside in the character of settlement, the immediate cause has been the elimination of fire.

The march of woody weeds was only a beginning. Biodiversity declined, and those creatures inimical to ranching or dependent on the old fire regime melted away like Arawaks before smallpox. Worse, the land started to erode. In southern Arizona a spectacular cycle of arroyo-cutting began in eerie lock-step to the cattle invasion. Elsewhere there were slides, debris flows, and garden-variety siltation on a bigger scale than ever known before. Alarmed, irrigation association campaigned for forest reserves and grazing control. (To oppose overgrazing was not to promote burning, however. Humus remained the guarantor of water.) The cumulative outcome was a colossal degradation of the landscape for which the tree, elsewhere a talisman of land health, was ironically an emblem of decay. By the early 1920s the declination had reached its nadir. Even cattle and sheep could no longer thrive on the land and had to be shipped to feed pens for fattening. Economically ranching depended for the most part on public subsidies, even as it remained a political power, that relationship not being entirely coincidental. The debate over the relative contributions of climate and anthropogenic activities continues but clear-eyed observers of the day had no doubt about the chain of causality. In 1933 Aldo Leopold wrote epigrammatically that "when the cattle came the grass went, the fires diminished, and erosion began."

By the time organized fire protection arrived it had only to confirm the fire ban announced by overgrazing. Fire suppression was, at first, an exercise in regional mopup. Aboriginal fires were sequestered onto reservations. Livestock had perverted fuels and quelled the impact of lightning fires; fires had little to burn, or they burned amid open forests, easily extinguished by pine boughs and blankets. Even ranchers that sought to "green up" spring pasture by burning found it difficult to do so. Arguments for "light burning" as the "Indian way" were dismissed by professional foresters as "Paiute forestry," and advocates were treated with the condescension normally reserved for perpetual-motion mechanics and circle-squarers.

But of course fire could never be abolished. Tremendous extents of the Southwest were committed to public reservations for Indians, forests, parks, the military, wildlife refuges, and other purposes. These lands remained quasi-natural, persisting in forms that would not yield to farm or city. Something like the native fuels endured, though often leavened with pyrophytic weeds. Lightning too continued its restless foraging, eager to seize whatever fuels came within its strike. Fire endured.

As decades rolled by, however, the fires that escaped now burned with unprecedented intensity and magnitude. Increasingly the saga of settlement moved from irony to tragedy. Fire had enhanced biodiversity; fire exclusion, through hoof and later shovel, destroyed it. Anthropogenic burning had improved fire control; fire suppression worsened it. Ranchers had sought to replace the wild with the domestic, and so they had done with grizzlies and cattle; but in the process they had also replaced the domesticated fire with the feral fire. If the land became less suitable for wild fauna, it became progressively more prone to wild fire.

That trend continued despite the New Deal's investment in conservation measures, including the Civilian Conservation Corps. Crown fires—large by virtue of their intensity as well as their size—increased from 10,127 acres per year in the 1940s to 15,117 acres per year in the 1980s, despite a massive commitment to high-tech firefighting. Fires that had rarely exceeded 3,000 acres in presettlement times now routinely reached 10,000 to 20,000 acres. It became apparent that to remove fire was as powerful an ecological act as to introduce it. In 1972 the Tall Timbers Research Station mustered a task force that singled out the pine forests of the Southwest as a case study in the consequences of fire exclusion. Slowly, grudgingly, the fire establishment admitted that its successes, ever more costly, were self-defeating. The tragedy of American fire history was not that wildfires were suppressed but that controlled fires were no longer set.

These concerns were not restricted to the Southwest, but when, some 25 years ago, the clamor for reform shook the national fire establishment, the Southwest quickly responded. For the new era this meant that somehow, in some form, preferably benign, fire had to be retained in the landscape. Where it had disappeared, it had to be restored. Prescribed fire for fuel reduction, for conversion of woodlands to pasture, for wilderness ecology, and for improved wildlife habitat became acceptable, if not commonplace. Excepting the South, the Southwest practiced more broadcast burning than any other region. But it was not enough.

Nowhere has anyone reintroduced fire as fully as it has been removed. Restoration failed to keep pace with even annual requirements, much less to make inroads in reducing a century's backlog of burning; those fires did little more than make the minimum payment on a credit card charged to its maximum limits. Often the fuel situation has worsened, particularly as logging exploded on the region's national forests in the 1980s and slash proliferated. It is easy to fund a dramatic fire fight; tough to justify the quiet burning which, if it is done properly, does not become a public spectacle. It was difficult to restore fire without restoring the other conditions that had helped sustain it. The end could only follow from the means. Somehow those accumulated fuels had to be disposed of.

The new Southwest is a product of the old Southwest. Those fuel loads on public lands are a kind of environmental debt, like toxic dumps, that will take decades of determined action to clean up; it is not clear that either the resolve or the money is there to do it; the backlog is too great, and the requisite social consensus too elusive. The logging of large trees inflames environmentalists. The removal of small trees does not suit the economics of sawmill logging. Above all, air quality considerations increasingly regulate the pattern of open burning. Woodsmoke must compete with industrial sources—smelters, coal-fired power plants, automobiles—for its share of the regional airshed. In 1975 and 1979 air pollution alerts in Phoenix resulted, in part, from an overload of broadcast burning on the Mogollon Rim. It is hard to explain to residents of the seventh largest metropolitan area that they can no longer burn fireplaces at will, but tens of thousands of acres need to be broadcast burned. The cultural distinctions make no difference to regional airsheds or to lungs degenerate with emphysema.

And now ex-urban sprawl plasters the private lands within and around the public domain with houses—another woody weed—adding all the problems of a suburban environment but with few of its correctives. Exurbanites are reclaiming a rural landscape but without a rural economy, and so increase the complexity (and expense) of fire protection without improving the prospects for fuel treatment, broadcast burning, or fire services. The developer is replacing the rancher, and summer homes and trailer parks and tourists, the throngs of sheep and cattle that once penetrated every meadow and forest paddock. A four-wheeled seasonal transhumance has replaced its four-hoofed predecessor. But the outcome for fire regimes remains unchanged. The fuel situation worsens, without a corresponding improvement in ignition.

Like shots fired in the dark, sooner or later lightning will hit the right combination of fuel, wind, and terrain with perhaps fatal effects, particularly when lightning discharges, as it does here, with the scatter of a shotgun blast. Increasingly it appears that wildfire is the only legally and politically acceptable form of burning, as though drive-by shootings were the only sanctioned form

of target practice. As drought seized the region in the late 1980s, wildfires increased. The Dude Creek fire was its climax, and its prophecy.

The character of Southwestern fire reflects the changing character of its human occupation. The classic Southwest blaze is a trying fire that exposes and assays, sometimes in dramatic fashion, the relationship between the natural landscape and the humans who live on it. It is difficult to reconstruct the impact of early humans, whose firesticks coincided with the colossal climatic fluctuations that ended the last ice age. Probably the Southwest featured a diffuse geography of burning in which firestick and grass created a regime that always simmered but rarely boiled, the intensity of fires following the tidal surges of rain and drought, the human firebrands smoothing out the jerky rhythms of lightning and wind, the very ubiquity of the burning helping to dampen catastrophic eruptions.

It is easier to document the dramatic alterations that have accompanied European settlement. The landscape mosaic became coarser, less able to absorb chronic disturbances. From some sites fire vanished—flooded into irrigable fields, paved into cityscapes, eaten away by livestock, or swatted out by determined fire crews. From others, fire receded temporarily, only to return in altered but reinvigorated form. Elsewhere it was kept in check only through ever-increasing investments in fire suppression. The slow growth rates in the semi-arid Southwest bought time; decades might pass before the consequences of fire exclusion became apparent or irreversible.

Now that interdependence is again shifting. Whatever climatic changes may occur, human-inspired change is outstripping it. Overall, human activity is increasing the total number of fires even as it shrinks to a razor's edge the border between a controlled fire and a wild fire. Smaller numbers of fires break free, but these rage over larger areas and with greater ferocity. Like the interest on a compounding debt, wildfire threatens to claim an ever larger proportion of the region's fire economy. The rural landscape that had once helped buffer between the urban and the wild continues to shrivel; in its place, houses insinuate into every nook and cranny of private land. There is little slack, small margin for error. The gradient between the wild and the urban steepens, building like an electrical charge. Eventually it will arc.

In Southern California, the intermix fire is a familiar morality play, almost a distinctive art form. Certainly the environment is built to burn. Clearly, the construction of expensive wood-shingled houses in mountains bristling with decadent chaparral and exposed to Santa Ana winds is an act of hubris. But the worst fires typically begin with arson. This transforms an environmental dilemma into a simple parable of human madness or malice.

It is more difficult to interpret fire in the Southwest, where lightning, not humans, normally supplies ignition. The relationships are more complex and balanced, the ironies more subtle. The tensions between nature and humans are multiple, not readily decoded into simple dialectics. The region remains a fire-baked mosaic of history and geography. It is not clear, for example, whether the Dude Creek fire was a freak event, the fiery manifestation of a record-shattering heat wave, or the calling card for a new era in which, regardless of technological investments, the pressure of human population and the legacy of suppressed fires will combine to make a truly ungovernable fire regime. Was it the old amalgam, merely intensified? Or is it a new compound, volatile as nitroglycerin, ready to explode at the first stumble? It seems to be both.

But neither is it obvious how to restore fire. However significant fire was to the Old Southwest, its removal was only one part of fashioning the New Southwest, and its reinstatement will not by itself restore that old landscape. It is not apparent by what means fire should be reinstated, nor to what ends it should be applied. Hardest of all is the determination of an environmental standard. The existential Earth offers no absolutes, only the record of many pasts and the prospects of many futures. The privileging of past peoples and landscapes speaks with no more authority. That anthropogenic fire has been an inextricable part of Southwest history does not declare how it might become a part of the Southwest's future. The one hard fact is that the existing scene had, for various reasons to various groups, become unattractive, unacceptable, and in the broadest sense increasingly uninhabitable. But however conceived, fire would remain, an inevitable end, an unavoidable means.

Fire belongs in the mountain Southwest, and unless the peaks flatten, the monsoon evaporates, the seasons homogenize, or the biota vanishes, those fires will continue. They ought to continue. The issue is how to relate to fire—how to keep it from destroying people and to how to keep people from transforming flame into a destroyer. Otherwise the border between the human and the natural will grind with greater and greater force, and out of that friction will come fire that no one wants and no one can control. The summer beacon will become a pyre on the mountain.

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Restoring synergistic human — wildlife systems: Managing ecosystem complexity for the 21st century

Dr. Thom Alcoze[1]

Abstract. The essential perspective presented in this session is that human beings must be considered as a functional component of our definition of ecosystem. This inclusion alters our historic approach to resource management decisions. The world view that human beings are part of the ecosystem or natural world rather than apart from nature represents a paradigm shift where the immediate concerns and needs of people do not predominate.

Historically, animal, plant, and other products of the natural world have been defined as gifts of nature or resources, free for the taking. Economically, this belief has represented great value to entrepreneurs. The harvest of natural resources has always been a vital and practical part of human history. Collecting the gifts of nature at no cost other than a person's labor and time for the purpose of exchange or barter at the market place is profitable and sometimes seen as a way to prevent the waste of useful commodities.

As a global society, we are now becoming aware that we consider more than basic economic factors. This awareness is a direct result of an ecological interpretation of nature where all components of an environment interact in complex patterns. As we define ourselves as part of the biosphere, we can begin to recognize a human niche that relies on a diverse set of factors that integrate subjective and objective ways of knowing.

Indigenous cultures of the Americas have dealt with many of the same issues that we are now beginning to address. Resource use strategies and technologies developed by First Nations people demonstrate a wide range of ecologically valid and economically effective methodologies for sustainable harvest. Many teachings that originate among the First Nations of the Americas illustrate how we can enhance our own perceptions about resource use to achieve a sustainable future.

Adaptive ecosystem restoration and management can represent a major diversion from historic attempts to interact with nature and natural resources. The perspective that nature can be managed and controlled according to human values and expectations has resulted in a resource crisis which must be addressed from a global or biosphere perspective. The significant reduction in the commercial availability of variety of natural resources due to overharvest is a clear indication that certain resource harvest practices must be altered. Future methods of natural resource harvest must successfully address the issue of sustainable resource use by finding ways to harvest natural resources and ensure that subsequent harvests of these resources will be equally productive. "Sustainable development" of natural resources has received much attention in the last few years. Natural resource professionals are beginning to address the issue of sustainable resource use through the development of innovative and often unique methods of harvest.

These new methods require changes in our perceptions regarding the relationship human beings have with natural resources and nature in general. Historically, animals, plants and other resources of the natural world have been viewed as gifts of nature, free for the taking. Collecting nature's gifts, the harvest of natural resources, has always been a vital and practical part of human history. This history, however, also demonstrates that the availability of natural resources is often finite.

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Resources can be exhausted, the extensive forests of North America are currently in a state of depletion which is in some aspects reminiscent of the elimination of a variety of wildlife species which occurred at the turn of the 20th century. As some species became less available, harvest pressure increased until populations could no longer successfully reproduce, leading to collapse or extinction. The extinction of the passenger pigeon in North America is only one example of the extinction of wildlife species directly related to overharvest. A belief that people are separate from nature and that nature represents an inexhaustible larder prevented a more careful and objective assessment of resource availability. Awareness of this fact could have reduced rather than increased harvest intensity. Renewable and nonrenewable resources are currently being consumed at rates that appear to threaten continued commercial harvest. It is imperative at this time to re-examine resource harvest practices and develop new approaches to the environment that promote long term use.

Economically, a "free gifts from nature assumption" represents great value to entrepreneurs. However, the view that resources can be acquired at no cost other than a persons labor and time is neither realistic or practical over a long period of time. When resources are used for the purpose of exchange or barter at a market place, profit rather than sustainable resource use controls harvest intensity. Rare resources can sometimes acquire inflated economic value increasing harvest pressure and further decreasing availability. The result of such a harvest strategy where infinite availability of resources is coupled with no cost harvest has historically resulted in depletion and or exhaustion of resource availability. As a global society, we are now becoming aware that to sustain the harvest of natural resources requires that we consider more than basic economic factors.

The restoration of human - wildlife systems is a contemporary expression of what has been referred to as "balance or harmony with nature". Native American Nations have been examined in this context to explore the possibility that America's First Nations are examples of living cultures that have retained a direct and interdependent *i.e.*, synergistic relationship with Nature. The validity of this idea has been questioned with the assertion that such concepts are either idealistic and simplistic romantic notions or an exaggeration of the level of technological sophistication and ecological awareness of Indian peoples.

The purpose of this presentation is to demonstrate that the harvest of natural resources by the original Nations of the Americas was in many ways superior to current multiple resource management principles. The application of this knowledge can be applied to the management of wildlife systems in ways that will promote the sustainable availability of natural resources through the 21st century.

The world view that human beings are a distinct aspect of the ecosystem or natural world rather than a component of nature represents a paradigm shift where the immediate concerns and needs of people do not predominate over the broader concern for a healthy natural environment. The inclusion and recognition of human beings as dependent on rather that independent of nature and natural resource availability can dramatically alter our approach to resource management decisions. This idea fits well with an ecological interpretation of nature where all components of an environment interact in complex interdependent patterns. As we define ourselves as part of the biosphere, we can begin to recognize a human niche that relies on a diverse set of factors that integrate subjective and objective ways of knowing to create sustainable resource harvest strategies.

Indigenous cultures of the Americas have successfully dealt with many of the same issues that are now beginning to be recognized as important to contemporary society. Resource use strategies and technologies developed by First Nations people demonstrate a wide range of ecologically valid and economically effective methods for sustainable harvest. Many teachings that originate among the First Nations of the Americas clearly illustrate how perceptions concerning resource use can be enhanced to achieve a sustainable future.

The Menominee Nation of Wisconsin have implemented a successful sustained yield forestry operation that may prove to be a model resource management strategy. The fundamental principles upon which the Menominee land use plan is patterned originate with the traditional values system of the "old ways". When these values are examined carefully, it is clear that the practices of sustainable development were, and continue to be, well understood by these woodland peoples.

The Micmac Nation of the northeastern woodlands of North America, also continue to practice sustained yield harvest. Based on resource partitioning among a widely distributed population, the Micmac use a family hunting territory system, which predates European contact. Using a rotation of harvest intensity to distribute the extraction of natural resources from the available surpluses of game, medicines, plant products, and other required commodities, resources are harvested over a long period of time from a number of harvest areas. As resource availability diminishes in one harvest area, the family hunting group relocates to other areas that have not been harvested for a sufficient period of time to allow the game and other resources to replenish.

Another example of sustained yield practice is found among the Beaver Nation of western Canada. The people of this modern day hunting society participate in

elaborate discussions among hunters prior to actual hunting forays. The purpose of these discussions is to accurately assess the most advantageous type(s) of game to be sought. In the case of moose, and other large mammals, a group of hunters will determine specific characteristics of the prey such as species, age, sex, and size class. This method of harvest, is an integral part of the hunting strategy of the Beaver Nation. They refer to these practices as "showing respect" for the game required for their survival by not taking more than necessary. Habitat modification was also an important part of the resource management strategies of many Native American Nations. The use of fire as an ecological tool by the Iroquois Nations has existed in the eastern woodlands for thousands of years. The controlled burns enhanced wildlife nesting habitats, browse production, soil nutrient content, and may be an example of early multiple resource management practice. Horticultural practices among Native American Nations demonstrate a high degree of sophistication and understanding about some of the specific requirements of plants that contributed to the success of agriculture in the Americas. Genetic hybridization of corn resulted in the development of a corn varieties that are adapted to regional environmental conditions. In the Southwest for example, corn varieties were created that are productive on less than 7 inches of rainfall per year. In the Northeast, corn is still grown in conjunction with bean and squash, an association called the Three Sisters by the Mohawk Nation. The high nitrogen demands of corn are compensated for by growing mounds of corn with pole beans, which as a legume fix nitrogen in the soil in sufficient quantities to replace that removed by the corn crop. The squash is used as a ground cover to inhibit light and thereby reduce the growth of other less desirable species.

Native American Nations represent an exceptionally successful, yet untapped resource of knowledge and information that illustrate some of the values and attitudes necessary to adjust lifestyles to the environment. The practices and beliefs of America's First Nations concerning the environment represent valid ways of knowing from which modern society can benefit. Rather than being interpreted as romantic notions, we desperately need to understand that the relationships between Native people and the environment are valid expressions of ecological principles that have a role in managing ecosystem complexity for the 21st century.

THE NEED FOR DIVERSITY AND ACCEPTABLE RISK

Richard Miller[1]

ABSTRACT. In the Southwest, minor variation in slope, elevation, aspect or soils can result in large differences in vegetative communities. These vegetative differences, in turn, provide habitats for a greater variety of wildlife. The value of maintaining diversity is widely accepted among conservation biologists and wildlife biologists and is even written into law as part of the National Forest Management Act. Southwestern ponderosa pine forests support a wide variety of mammal and bird species, each of which uses and benefits from a portion of the diversity in ponderosa pine forests. Each species occupies a niche made up of several to many habitat components. Each of these species prospers or declines in response to changing habitat conditions as determined by its needs and flexibility. By providing diverse vegetative conditions and habitats we can insure diversity of wildlife species, and if current ecological theories are correct, insure greater stability over time.

By providing for the habitat needs of native wildlife species, there will be a risk of some avoidable losses to fire and insects. Finding acceptable levels of risk is both difficult and controversial. In order to provide some wildlife habitat needs we must include some areas with tree densities which make forest managers nervous. If we reduce risk from bark beetles and fire to low levels we then increase risks of local or general loss of some wildlife diversity. It is not simple to manage for diversity, and it sometimes gets swept aside in our eagerness for simplicity in management paradigms. Current thoughts about presettlement conditions should be included in a mix of conditions within future forests. No paradigm should be applied as a blanket across southwestern ponderosa pine forests whether that paradigm be based on our understanding of spotted owls, or based on our understanding of presettlement forests. Future forest management should include more open conditions than have been emphasized in the recent past but it must retain a range of both density and size of functional areas or run unacceptable risks of losing diversity and stability.

INTRODUCTION

In the Southwest minor variation in slope, elevation, aspect or soils can result in large differences in vegetative communities. These vegetative differences, in turn, provide habitats for wildlife. Southwestern ponderosa pine forests support a wide variety of mammal and bird species, each of which uses and benefits from a portion of the diversity in ponderosa pine forests. Each of these species prospers or declines in respond to changing habitat conditions as determined by its needs and flexibility. By providing diverse vegetative conditions and habitats we can insure diversity of wildlife species, and if current ecological theories are correct, insure greater stability over time. The value of maintaining diversity is widely accepted among conservation biologists and wildlife biologists and is even written into law as part of the National Forest Management Act.

It is not simple to manage for diversity, and it sometimes gets swept aside in our eagerness for simplicity in management paradigms. Current thoughts about presettlement conditions should be included in a mix of conditions within future forests. But, no paradigm should be applied as a blanket across southwestern ponderosa pine forests whether that paradigm be based on our understanding of spotted owls, or based on our understanding of presettlement forests. Future forest management should include more open conditions than have

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been emphasized in the recent past but it must retain a range of both density and size of functional areas or run unacceptable risks of losing diversity and stability.

Changing paradigms

Recent history of forestry paradigms in Arizona

Several major patterns of thoughts or paradigms have directed forest management in Arizona over the last one hundred years. Managers in the past have tended to focus on trees that were removed and on economic value of the wood. Today's paradigms has shifted to "focus on what is retained rather than what is removed", and to "arrange structures on a landscape in a manner that restores and maintains landscape integrity" (O'Hara et. al. 1994). These new paradigms attempt to balance commodity production with forest health in the broadest sense and with multiple uses on a large or very large scale. Interest in a part of the new paradigm which emphasizes restoration of presettlement forests had led to this conference.

A variety of harvesting practices have been used to implement these dominant paradigms. Something similar to today' seed cuts or clear cuts were originally used on entire sections. By 1904 75,000 acres of what is now the Coconino National Forest had already had 95 to 100% of the timber volume removed (Lieberg et. al. 1904). Harvesting first changed to leave some residual trees and then later moved to a practice of even aged management and overstory removal. For instance a 1965 plan called for removing all trees over 28.5 inches in diameter (USDA Forest service 1965). One of the paradigms that arose as an imperative was the need to capture mortality. The idea was simply not to let trees go to waste by dying before they could be harvested. We have learned that capturing all the mortality has negative effects on birds, bats, small mammals and insects (DellaSala et al. 1995).

Harvesting methods after the second world war leaned towards variations in single tree selection still stressing capturing mortality. In the 1970 and 1980s harvesting turned to even-age management that in some ways resembled what had been done one hundred years before but with smaller scale cuts units.

Our paradigms are the sources of many of our attitudes and behaviors (Covey 1989). When paradigms change, the behavior of individuals and organizations react. One predictable reaction is to assume that none of the insight or knowledge gained through the old para-digm has real worth. Another equally predictable but opposite reaction is absolute defense of the old paradigm. Only rarely are new paradigms accepted as a useful new thought without one or the other of these two reactions entering in.

While writing this paper I had to reconsider my own assumptions about paradigms and presettlement forests. A list of assumptions follows because they influence the conclusions which I will present later. I am assuming:

- There is truth in the new paradigm. There are generally more trees now, and fewer old large trees. The new thoughts deserve application in management.
- There is both truth and continuing reality in the old paradigms. A great many acres are dedicated to even aged regularly spaced stands designed to produce near optimum yields of lumber. We will have to manage with those stands in that condition or a derivative of it for a long time.
- There is value in realizing what ever we do is an experiment, because there is so much we do not know. Paradigm change will come again as we try new ideas, learn and experiment.
- We can use the past for a guide point, but we can not return to the past. Changes have occurred in soils, water, forests, people and time scales. The time scale change maybe most important. Nature could afford to wait 20 to 60 or more years for regeneration events or other "needed" changes, but man will not wait.

The need for diversity

In their 1991 position on biological diversity in forested ecosystems the Society of American Foresters defined biological diversity as follows:

> "Biological diversity refers to the variety and abundance of species, their genetic composition, and the communities, ecosystems and landscapes in which they occur. It also refers the ecological structures, functions, and processes at all these levels. Biological diversity occurs at spatial scales that range from local through regional to global."

The position goes on to explain:

> "Human activities that threaten biological diversity are largely caused by economic development and by the adoption of simplified production systems that specialize in a single plans or animal species. The challenge for foresters is to balance economic development and biological diversity."

Maintaining diversity requires us to strike this balance on multiple scales of area and time. We must at the same time provide for: a mouse that lives only a year and a half and most of that on a quarter acre, a bear which lives perhaps ten years and has requirements at the landscape scale, and trees which lives hundreds of years but respond strongly to microsites when they are

seedlings. Each plant and animal has requirements which must be met in order for the species to remain part of the local diversity. For many of the animals these requirements are different for different activities.

For many of the animals, habitat requirements change with season and different activities. We can use turkey as an example. In summer, turkey forage in small openings (Hoffman et al. 1993). During the day, turkey rest in small patches of relatively dense young timber which has a down log or other low perch within the patch (Hoffman et al. 1993). At night turkeys roost in large, usually old ponderosa pines with an open limb structure (Hoffman et al. 1993, Phillips 1980). Turkey do best where springs and seeps are common because of the abundance of lush vegetation and insects, which benefit poults (Hoffman et al. 1993). Turkey poult survival is linked to height of ground cover and therefore is affected by grazing. In addition turkey have a separate sets of requirements of winter range and nest sites (Hoffman et al. 1993). In order for turkey to do well all of these elements must be present in close proximity to each other (Hoffman et al. 1993, Wakeling 1993).

According to the journals of early anglo explorers turkey were apparently abundant in presettlement times (Davis 1973). The abundance of turkey implies that all their needs were being met by presettlement conditions. However, if we could now return to any of the current concepts of presettlement forest structure, turkeys probably would not fair well because other landscape scale changes have occurred.

The City of Flagstaff and other area communities have built over a number of local springs removing them from available turkey habitat. Other springs and seeps have dried up as a result of changing hydraulics, these losses again changed the quality of turkey habitat. Grazing now impacts the herbaceous ground cover needed by turkeys. And perhaps most significant, human activity in the forest has increased dramatically influencing the effectiveness of remaining habitat.

In Game Management Unit 6A which lies south of Flagstaff, the Forest Service estimates 1,600,000 Recreation Visitor Days per year. To put the impact implied by 1,600,000 visitor days in perspective, one study near Flagstaff found that 38% of the vehicles that noticed a turkey decoy along a road stopped and shot at the turkey decoy (Jones and Barsch 1993). Both direct loss of turkey to poaching and indirect loss because of reduced habitat effectiveness are very likely in effect when recreation use is this high.

People pressure and habitat changes can both reduce the usefulness of turkey habitat (Hoffman et al. 1993) and may make some areas into nonhabitat. Any action which reduces the hiding cover available in Unit 6A will almost certainly affect the turkey population as the loss of cover increases the affects of people. It is possible, given the high recreation pressure, that the turkey population in Unit 6A could be eliminated by a reduction in cover. If we want to keep turkey as part of the diversity in Unit 6A, cover will have to be maintained, whether or not we can agree that the cover was part of presettlement conditions.

Other animals could have been used as an example in place of the turkey. I could have used the example of tassel eared squirrels and their mutual beneficial relationship with fungi and the ponderosa pine. A key link in this relationship being the tree densities needed to provide the shade and moisture required by the fungi. The example could have been the whitetail deer which live in the canyons along the Mogollon Rim and their need for the diversity of vegetation in those canyons. Another example could be the black bear which are very restricted to dense vegetation and the diversity in the canyons. I have been using game animals as examples since our information is often better, and also because of the fireworks set off when nongame animals like the spotted owl or goshawk are mentioned.

Presettlement diversity

There is evidence for a diversity of structure and density in presettlement forests. Woolsey (1911) noted, "In places it (the ponderosa pine forest) is made up of blackjack, with an occasional mature yellow pine fast declining in vigor. In others there may be an old mature stand of veterans with complete reproduction beneath. On the limestone formations, with deep soil, the stand is usually more thrifty than on lava." He also called stands on the Coconino "frequently very dense," and recorded 27,352 to 33,055 BF/acre in some plots and 201,641 BF/acre for a 1/23 acre plot on the Zuni National Forest. These would be extremely dense stands. Plummer (1904) also remarked on the "exceptionally heavy stands of young timber" in some townships.

In discussion the use of their research Covington and Moore (in Arizona Game and Fish Department 1993, appendix 2) state, "...the results point to the extreme variation from point to point within one area and from landscape to landscape across a region." At 8 plots at Bar-M Canyon, presettlement timber volumes ranged from 341 BF/acres to 10,381 BF/acres.

Wheeler (1878) wrote of the White Mountains as a "densely timbered range." Wheeler again refers to a dense growth of timber between Camp Tulerosa and Camp Verde. Wheeler also refers to the "dense timber growth, with here and there fertile valleys and open glades" near the San Francisco Mountains. The expeditions chronicled by Wheeler were commissioned by the United States to report on conditions in the west specifically including vegetation and deserve credibility.

In appendix H1 (preliminary Botanical Report) to the Wheeler series of reports is a striking example which refers to both dense forests and open park like stands within a few sentences of each other. J. T. Rothrock a surgeon to the party who was also acting as a naturalist as was then common, writes of crossing the Zuni Mountains thus "Our ascent lay through dense forests of pine and fir". Two sentences later Rothrock describes a summit above Fort Wingate as "a fine open park-like region, with a large growth of yellow pine and fir covering the hill sides". It is very likely that J. T. Rothrock intended these different descriptions to covey that he had traveled through two different forest conditions.

Edgar A. Mearns writing in 1890 speaks of the San Francisco Mountains and the Mogollon Range as "well wooded" and "adorned with many beautiful parks and elevated valleys". Later he writes of the Pinal range "whose north slopes are heavy timbered". Mearns describes ridges north of the Sierra Prieta as "heavily timbered". Again I believe these description were intended to convey to the reader a difference in forest conditions.

I do not dispute that many more young trees and many fewer old trees are now present across much of the landscape than were present in presettlement times. However, it is also important to note that apparently conditions were not uniform in presettlement times. Variety in the mix of structures and tree densities would benefit wildlife and replicate some conditions which, although they may not have been dominant, apparently were present in presettlement forests.

The risk involved in forest management

Numbers of authors have written articles and books on why we should care if diversity is at risk (for example, Noss and Cooperrider 1994, Hunter 1990) . The retired chairman of the subcommittee on the Environment, U. S. House of Representatives, James H. Scheuer (1993) put one of the best reasons well when he wrote:

"Each endangered species is not an isolated species but a part of an ecosystem. Eliminating one part of an ecosystem will have adverse effects on the whole system for the whole is indeed greater than the sum of its parts. Our ecosystems depend upon the species connecting within them for their stability and resilience".

If the biological and ethical reasons for attempting to maintain diversity are not enough then we should remember that the National Forest Management Act requires the maintenance of the diversity of plant and animal communities on National Forests.

There is risk to diversity each time we change paradigms. That risk increases the more thoroughly and rapidly we apply the new paradigm. Each plant and animals has requirements that should be maintained as part of a mix of conditions in the forest if we wish to maintain them as part of the area's diversity. Most often these requirements are only partially known. As illustrated with turkey cover, known habitat requirements include conditions which are not in the range of some proposed management. There are at least three possibilities for each of these plants and animals:

1. It is possible that the assessment of presettlement conditions is correct and these animals were able to exist in those conditions.
2. It is possible that some current management proposals do not yet contain the full range of conditions which existed in presettlement times, and some animals could not tolerate being limited to a narrow range of management conditions.
3. Finally it is possible that some or all of these animals could have survived in presettlement times under conditions which they could not now tolerate. Possible reasons for the lack of tolerance could include: the increased people pressures, the relative lack of old trees, reduced availability of water and its associated riparian vegetation, or any of the other factors we have changed.

If either of two of the three possibilities are correct blanket application of any prescription across the landscape will be a mistake. The risk to diversity lies in the potential failure to include in our range of desired conditions and the known and unknown habitat conditions needed to maintain the entire range of wildlife and plant species.

There is risk to forest health from pests and diseases. These risks can be related to stress which in turn can be partially linked to tree density. Larson et al (1983) after studying tree vigor and pitch tubes from beetle attack recommended stocking at less than 34 square meters of basal area per hectare (148 sq ft of basal area/acre). Larson et al. (1983) cited two other studies which had reach similar conclusions. At this level of stocking the authors felt most trees could withstand at least a moderate attack by bark beetles. Risks of beetle attack and some other pests can be greatly reduced by maintaining very open forests.

For some diseases such as the armillaria root rot tree harvest facilitates the spread of the disease. How best to reduce risk in these situations is as yet largely unanswered question. For mistletoe the standard treatments have been density reduction harvests or salvage sales aimed at removal of most or all the diseased trees. But tree harvest can in fact increase the rate of spread of mistletoe under some conditions (Fairweather and Frank, 1992).

In discussions of forest health it is necessary to differentiate between health of the individual trees and ecosystem health. The plant and insect species we refer to as forest pests have an important role to play in the forest ecosystem and can help to maintain ecosystem processes (DellaSala et al 1995). Some risk to individual trees is reasonable and desirable to maintain the forest ecosystem.

The risk of crown fire are apparently also related to tree densities. Large areas of homogenous dense forest present what most people would agree, is an unacceptable risk of crown fires. There is, however, some evidence that timber harvest may not be effective in reducing landscape scale risks of catastrophic fires (DellaSala et al 1995). Under most but not all weather conditions the risk of crown fire being out of control can be reduced by use of fire breaks. Fire breaks can be areas of low density, presettlement forest interspersed throughout an area. In areas of highest risk to people and property it makes sense to reduce the risk of crown fires to a minimum. But fires, even crown fires, produce some habitat conditions, such as aspen regeneration or stands of shrubs, not replicated by other management activities (DellaSala et al 1995). Some risk of crown fire can be accepted and indeed must be accepted, since under extreme weather conditions crown fires are possible even in thinned forests.

The same insects that are pests on trees are food for woodpeckers and other birds. Mistletoe provides nests sites for birds and there is some information suggesting higher bird diversity in stands affected with mistletoe. Fires can provide groups of snags and new shrub habitat. The question is not one of eliminating an avoidable risk but rather of balancing the risks of loss of wood fiber and property with the risks created by efforts to minimize risk of fire or insects and diseases.

People, especially in large numbers, complicate both management and risk assessment. Campers and smokers increase risks of fire. Large numbers of people of any type reduce habitat effectiveness and increase the risk of loss of diversity. Large numbers of people may also increase risk tree disease by increasing the injury rate of trees.

The limitation of our knowledge impose definite risks on our decisions (Noss and Cooperrider 1994). Our knowledge of forested systems and how to manage them is progressing rapidly. Not that many years ago there was not even a field of study called restoration ecology. Just a few years ago we did not know of the tassel eared squirrel relationship with fungi and ponderosa. Three years ago we did not know bats were using snags in the ponderosa pine forest except from anecdotes. A lack of knowledge is not cause for inaction from fear of making mistakes, it is however a very good reason to proceed with some caution and respect for what we don't know.

Suggestions for implementing a new paradigm

In discussing way to maintain diversity in forested systems several authors (Balda 1975, DellaSala et al. 1995, Hunter M.L. 1990, Schoen et. al. 1981) have suggested maintaining a range of forest structure and density and/or a range of block sizes. I believe the idea of maintaining both a range of structure and density, and a range of block sizes has promise in the ponderosa pine forests of the southwest. A management philosophy incorporating a range of stand density and structure and a range of block sizes allows managers to take action while conserving options and would appear to be our best course for the future. In order to conserve options and maintain diversity for the future I suggest maintaining nearly the full range of current conditions, excluding only the most dense stand conditions. I also support creating areas of presettlement restoration in a range of block sizes within a matrix of block sizes and other stand conditions, and monitoring their use by wildlife. I do not recommend maintaining the current proportions of density classes or their current distribution on the landscape.

In implementing any new paradigm we cannot totally avoid risk but we can manage risk and balance different risks. For instance creating areas of presettlement conditions could be used to reduce risk of fire around cities such as Flagstaff. In fact this is already being proposed on the Coconino National Forest adjacent to Flagstaff. The presettlement like prescription is proposed on lands which pose the greatest wild fire threat to the City of Flagstaff. That same proposal also includes two other prescription which will be leave a greater residual tree densities. These prescriptions were proposed to meet recreation and wildlife objectives where fire risk to the city is less.

We can accept risks on portion of the forest that we might not want to accept across the whole forest. As an example, using the work of Larson et al (1983). It seems reasonable retain part of the forest at densities approaching his maximum. In these areas we would accept what Larson et al felt were acceptable risks of increased bark beetle mortality and would be providing better conditions for the false truffle fungi, Abert's squirrel, turkey, and black bear among other species.

The well known quote attributed to Aldo Leopold, "The first rule of intelligent tinkering is to save all the pieces" should remain one of our management principles. Maintaining diversity in stand age, condition and density as well as block size equates to keeping all the pieces.

Implementing **any** single idea uniformly across landscapes places future managers in a position with very few, if any, options. The thinner the trees in the

residual stands after treatment the fewer the management options until the original stand has matured. Under Integrated stand management we jointly committed thousands of acres to even aged management in 30 to 100 acre blocks. If we could do those acres over again most of us would manage them very differently now.

We should remember the lessons we have already learned and use them to better implement the new paradigms. Future forest management should include more the open forest conditions currently described as presettlement, but it must retain a range of tree density, stand structure, and size of functional areas or run unacceptable risks of losing diversity and stability.

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Historical perspectives on forest insects and pathogens in the Southwest: Implications for restoration of ponderosa pine and mixed conifer forests

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Abstract. Forest insects and pathogens have affected southwestern pine and mixed conifer forest ecosystems for millennia. The majority of these organisms affecting our forests today are native and have co-evolved with their hosts. These agents function directly as herbivores, and along with fire, are among the major disturbance agents affecting southwestern forests. There is evidence suggesting that as the incidence and severity of fire has changed here so has the nature of forest insect and pathogen activity. These changes in disturbance patterns reflect changes in vegetation composition, structure and density. These changes in turn have many repercussions for restoration of pine and mixed conifer landscapes today. The introduction of exotic agents, such as white pine blister rust, will also significantly affect southwestern forests. Long term success of ecological restoration of pine and mixed conifer landscapes in the Southwest will depend in part upon how well we incorporate into management our knowledge of how forest insects and pathogens affect ecological processes and functions.

Introduction

Forest insects and pathogens have affected southwestern pine and mixed conifer forest ecosystems in the Southwest for millennia. The majority of these organisms affecting our forests today are native and have co-evolved with their hosts. Among the major insects and pathogens affecting pine and mixed conifer forests today are several species of bark beetles, defoliators, dwarf mistletoes, and root pathogens. These agents function directly as herbivores, and along with fire, are among the major disturbance agents affecting southwestern forests.

There is some evidence suggesting that as the incidence and severity of fire has changed here in the Southwest so has the nature of forest insect and pathogen activity. In some cases there have been changes in the spatial and temporal patterns of outbreaks while in other cases the potential for more severe outbreaks has increased. These actual and potential changes in disturbance patterns reflect changes in vegetation composition, structure and density. In this paper we will explore some this evidence. These in turn have many repercussions for restoration of pine and mixed conifer landscapes today. The introduction of exotic agents, such as white pine blister rust, will also significantly affect southwestern forests. Long term success of ecological restoration of pine and mixed conifer landscapes in the Southwest will depend in part upon how well we incorporate into management our knowledge of how forest insects and pathogens affect ecological processes and functions.

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History of outbreaks

Bark beetles

Numerous species of bark beetles affect pine and mixed conifer forests in the Southwest, attacking all species of coniferous trees. Most are fairly host specific, and are confined to primarily one tree species. Some of the more important ones include: the mountain pine beetle, *Dendroctonus ponderosae* Hopkins, in ponderosa and white pines; western pine beetle, *D. brevicomis* LeConte, in ponderosa pine; roundheaded pine beetle,

D. adjunctus Blandford in ponderosa pine and Chihuahua pine; pine engraver, *Ips pini* (Say), and the Arizona five spined ips, *I. lecontei* Swaine, in ponderosa pine, *D. pseudotsugae* Hopkins, the Douglas-fir beetle, in Douglas-fir; and *Scolytus ventralis* LiConte, the fir engraver, in white fir; (Wood 1982). This paper will discuss the first three of these species.

Of these species the most extensive historical record exists for the mountain pine beetle from the Kaibab plateau in northern Arizona. From this area, there is evidence of numerous mountain pine beetle outbreaks dating back to the 1800's with outbreaks occurring approximately every 20 years. Prior to 1917, dates of outbreaks were estimated by entomologists using increment cores and evidence of pitch pockets from unsuccessful attacks. Blackman (1931) estimated that outbreaks occurred during the following years based on this evidence: 1837-1846, 1853-1864, 1878-1892, 1886-1892, 1906-1910.

The first outbreak that was extensively studied occurred between 1917 and 1926, and was caused primarily by the mountain pine beetle, referred to then as the Black Hills beetle, (Blackman 1931). In this outbreak, he reports that 12 percent of the ponderosa pines were killed on the plateau both on the Kaibab National Forest and Grand Canyon National Park. The peak of the epidemic occurred in 1921 and 1922 when very extensive mortality occurred on Moquitch Ridge, Castle Ridge and other areas. During these two years it was estimated that between 225,000 and 275,000 trees were killed.

Since 1926, four additional outbreaks have occurred on the plateau, 1935-1938, 1950, 1973-1980 and most recently a small outbreak developed beginning in 1992. Parker and Stevens (1979) report that the 1935 and 1950 outbreaks were localized. The outbreak in the 70's began in 1973 in the East Lake area and at its greatest extent in 1976 and 1977 scattered pockets of mortality were scattered across approximately 75,000 acres according to Region 3 aerial detection survey records.

In addition to information on patterns of outbreak frequency and severity, the other important component to consider is the susceptibility of the host type to mountain pine beetle. Comparisons of stand conditions from 1910, based on the Lang and Stewart inventory, with those of today indicate that pine forests on the plateau have changed tremendously (Ellenwood personal communication). Forests of today have many more trees per acre, and higher basal areas than those at the turn of the century. At the same time the susceptibility of these forests to mountain pine beetle has also increased. At the present time we estimate that 32 percent of the pine type on the plateau is in a low hazard condition, 35 percent in a moderate condition and 33 percent in a high hazard condition.

Elsewhere in the Southwest, reports of historic beetle activity, particularly outbreaks, are scarce. Hopkins (1909) reports that in general the amount of tree mortality caused by what he called the Black Hills beetle, now the mountain pine beetle, was less in New Mexico and Arizona and southern Colorado than in the Black Hills. He also reports that the western pine beetle, which was then considered to occur along the Pacific coast from California to Washington and Idaho, caused more mortality than what he called the southwestern pine beetle, now synonomized, occurring in the Southwest. Woolsey (1911) reports that on a sale of approximately 30,000,000 feet of timber on the Coconino National Forest, fully 10 percent of the standing trees were dead. He attributed this to drought, but notes that drought weakens trees and predisposes them to insects. It is possible that some or most of this mortality could been caused by bark beetles in association with drought. Pearson (1950) reports bark beetles to be among the four main causes of mortality and notes that they are a major cause of death in reserve stands, causing about one third of the mortality reported by all causes. In virgin stands monitored between 1925 and 1940, he notes that they accounted for 1.6 percent of the mortality to trees overall, with somewhat higher rates for trees in the larger diameter classes. Meanwhile in cutover stands they accounted for 0.3 percent of the mortality to trees.

In recent times, however, some large bark beetle outbreaks have occurred. Perhaps the most notable ones have been in the Sacramento Mountains of southeastern New Mexico. Prior to the 1970's, outbreaks in the Sacramentos were small, a few thousand acres at most (Massey et al 1977). However, since 1971 two large outbreaks have occurred. In the early 70's, an estimated 400,000 trees were killed on over 150,000 acres (Massey et al 1977). This outbreak, which occurred in second growth ponderosa pine, resulted in mortality to between approximately 11 and 54 percent of ponderosa pines in sampled stands (Stevens and Flake 1974). Meanwhile, the average diameter of ponderosa pine remained about the same. Overall the outbreak resulted in a shift from ponderosa pine to other species such as Douglas-fir, white fir, southwestern white pine, pinyon, juniper, oak and aspen. Beginning in 1990 and extending to the present another outbreak involving both the roundheaded pine beetle and the western pine beetle killed an estimated 133,000 trees over some 87,000 acres.

Around the same time, two smaller yet significant outbreaks of roundheaded pine beetle have occurred in the Pinaleno Mountains of southeastern Arizona, one starting in 1964 and covering about 640 acres, and the second starting around 1988 with larger groups of mortality appearing starting in 1991 (Flake 1970, Wilson 1993). The recent outbreak has affected about 2,130 acres of pine and mixed pine forest type. We are not

aware of any sizable outbreaks prior to this time in the Pinalenos. We expect that this trend will continue and may extend to other areas in the Southwest, primarily due to increasing tree densities as compared to prevailing conditions present prior to European settlement (Johnson 1995).

Western spruce budworm

Fire exclusion, grazing, and past logging have had a great effect on southwestern forests by changing stand structure and species composition, resulting in a net increase in the acres of the mixed conifer forest type (Johnson 1995). These changes have also affected the nature of western spruce budworm outbreaks. This insect is the major defoliator of white fir and Douglas-fir within the mixed conifer type in this region.

An excellent chronology of western spruce budworm (WSB), *Choristoneura occidentalis* Freeman, outbreaks for southern Colorado and northern New Mexico has been assembled by Swetnam and Lynch (1989 and 1993). Western spruce budworm outbreaks have occurred in this region at irregular intervals in mixed conifer forests for at least the last 300 years. At least nine regional outbreaks have been identified in the mixed conifer stands of the Colorado Front Range and Sangre de Cristo Mountains between 1690 and 1989 based on tree ring studies.

The results of these studies indicate that a change has occurred in the pattern of budworm outbreaks during this century. Though the frequency of moderate to severe outbreaks during this century is not clearly more or less than during previous centuries, the spatial and temporal pattern of occurrence has changed (Swetnam and Lynch 1989). Of the past three centuries the 20th century had the longest intervals of reduced budworm activity (Swetnam and Lynch 1993). Outbreaks in the latter half of this century have become more synchronous over the host type suggesting that recent outbreaks have become more extensive than previous outbreaks (Swetnam and Lynch 1989). There was also evidence suggesting that the most recent outbreak has been more severe than past ones, resulting in the largest mean growth reduction (Swetnam and Lynch 1993). This change in spatial and temporal pattern and severity of budworm outbreaks has been linked to changes in age structure and species composition, involving widespread establishment of younger multi-storied, shade-tolerant, budworm host susceptible trees resulting from management practices around the turn of the century.

Dwarf mistletoes

The dwarf mistletoes (*Arceuthobium* spp.) have evolved with their hosts, as evidenced by fossil records dating back to the Pleistocene epoch (Hawksworth and Wiens 1972). More than 2 million acres of National Forest lands in Arizona and New Mexico are infested with dwarf mistletoes (Johnson and Hawksworth 1985). Most southwestern conifers are parasitized by species of *Arceuthobium*, however, the most significant damage occurs to ponderosa pine infected with the southwestern dwarf mistletoe (SWDM), *A. vaginatum* subsp. *cryptopodum* (Engelm.) Hawksw. & Wiens) and Douglas-fir infected with the Douglas-fir dwarf mistletoe, *A. douglasii* Engelm.

Little historical information concerning the distribution and severity of dwarf mistletoe in pine and mixed conifer forests in the southwest exists. Woolsey (1911) reports in his paper on western yellow pine in Arizona and New Mexico that between 1 and 2 percent of all western yellow pine (ponderosa pine) are attacked. He also notes that areas of heavy infection levels exist on the Coconino and Tusayen National Forests.

The distribution and rate of increase in dwarf mistletoe populations are affected by numerous host, stand and environmental factors including: site quality, host vigor, host age, stand density, stand structure, stand composition, and stand history (Parmeter 1978). Wildfires are one of the primary ecological factors in determining the distribution and intensity of dwarf mistletoes in unmanaged coniferous forests (Alexander and Hawksworth 1976). Relatively complete burns tend to have a sanitizing effect on infected stands, while partial burns can lead to rapid infection of regeneration if scattered infected trees remain following the fire. Fire scar chronologies from southwestern forests for the period from 1700 to 1900 indicate mean fire intervals of 4 to 5 years for ponderosa pine sites and 6 to 10 years for mixed conifer sites (Swetnam 1990). Since severe dwarf mistletoe infection leads to accumulations of dead trees, witches' brooms, and other fuels, the frequent low-intensity fires common in pre-European settlement forests probably reduced dwarf mistletoe in many areas (Parmeter 1978). Surveys of the ponderosa pine forests in Arizona and New Mexico conducted in the 1950's and 1980's (Maffei and Beatty 1988) suggest that the incidence of southwestern dwarf mistletoe may have increased due to human activities such as selective harvesting practices and suppression of wildfires.

Although the basics of dwarf mistletoe control have been known for a long time (Koristian and Long 1922, Pearson 1950), past cutting practices may have exacerbated SWDM infection in southwestern ponderosa pine stands. Light improvement selection cutting was extensively practiced throughout the Southwest until the 1980's (Heidmann 1983). Under this system, mortality losses in virgin stands were reduced by harvesting merchantable trees that were dying or expected to die during the following 20-year cutting cycle. A long-term

study of silvicultural control of SWDM on the Fort Valley Experimental Forest, near Flagstaff, AZ (Heidmann 1983) compared the effects of light improvement selection, limited control, and complete control in heavily infected mature ponderosa pine stands. After 27 years, the only effective silvicultural control method was complete removal of infected overstory and understory trees.

Root diseases

Root diseases caused by *Armillaria* spp. (Fr.:Fr.) Staude, *Heterobasidion annosum* (Fr.) Bref., and *Inonotus tomentosus* (Fr.:Fr.) S. Teng are common in many mixed conifer, spruce-fir and some pine stands throughout the Southwest. A survey of commercial timber-producing lands on six National Forests in Arizona and New Mexico indicated that root diseases and associated pests were responsible for about 34 percent of the trees killed (Wood 1983). After a half century of fire exclusion and selective harvesting, the incidence of root disease is suspected to have increased in mixed conifer stands in the Intermountain Northwest as ponderosa pine, a species that is relatively insect and disease resistant, is replaced by Douglas-fir and true firs, species that are much more prone to infection by root diseases (Hagle and Goheen 1988). Similar shifts in species composition are occurring in southwestern mixed conifer stands (Swetnam and Lynch 1989). Even though direct comparisons of root disease incidence in pre-European settlement and present times are not possible, these diseases have probably increased in southwestern forests due to the greater abundance of susceptible hosts and inoculum created by harvesting.

White pine blister rust

White pine blister rust is caused by the exotic fungus *Cronartium ribicola* J.C. Fisch. This fungus has spread throughout virtually the entire range of western white pine since its introduction to British Columbia in 1910. It was discovered in the Southwest for the first time in March 1990 on southwestern white pine in the Sacramento Mountains near Cloudcroft, New Mexico (Hawksworth 1990). Surveys indicate that this disease is now present throughout most of the range of southwestern white pine on the Lincoln National Forest and the adjacent Mescalero Indian Reservation (Hawksworth and Conklin 1990). The fungus has caused seedling and sapling mortality, as well as extensive branch mortality on all size classes of southwestern white pine in the affected areas. This disease will likely have a major impact on the white pine population in the Sacramento Mountains. Young trees will suffer more damage than larger, older trees since they are more prone to girdling cankers. The gradual decline in southwestern white pine regeneration will have significant impacts on the species diversity of the mixed conifer forests in the Sacramento Mountains. This disease can also be spread, either by man's activities or by windblown spores, to other areas of southwestern white pine, limber pine, and bristlecone pine in the Southwest.

Implications for Restoration of Pine and Mixed Conifer Landscapes

Prior to European settlement, disturbance events in pine and mixed conifer forests were frequent and of low intensity, predominated by the effects of fire (Cooper 1960, Covington et al. 1994). Since European settlement, the patterns of disturbance have changed and have resulted in different forest and insect and disease conditions (Covington et al. 1994, Swetnam and Lynch 1989, USDA Forest Service 1995).

In the ponderosa pine type, these changes have resulted in higher densities, and canopy closures of ponderosa pine (Covington and Moore 1992, Covington et al 1994). At the same time fire frequencies have decreased and fire size and intensity has increased. Both of these changes, along with past management practices, may have resulted in an increase in dwarf mistletoe incidence and severity. Many of these conditions also favor bark beetle outbreaks, and similar to fire, beetle outbreaks in the future may become larger and more intense. Larger outbreaks could result in dramatic changes in current forest structure, composition, and function, including creation of openings, depletion of larger diameter trees, and an increase in fire hazard due to the buildup of standing and down woody material.

In the mixed conifer cover type, dense, multi-storied, second growth stands predominated by Douglas-fir and white fir, have replaced more open stands composed of these species and a significant component of ponderosa pine, as a result of selective harvesting, and fire exclusion (Covington et al. 1994, Swetnam and Lynch 1989). These stands are now very susceptible to both western spruce budworm as well as root disease. There is evidence that budworm outbreaks are becoming more extensive and more severe. As with the bark beetles, an increase in activity of these agents is likely to increase fire hazard in mixed conifer landscapes.

The introduction of the exotic white pine blister rust fungus into southern New Mexico will have significant impacts on the biodiversity of mixed conifer forests since southwestern white pine shows little innate resistance to this pathogen. Further, since this species is not a host of western spruce budworm and is more resistant to many

root diseases, its decline in mixed conifer forests in the Sacramento Mountains will favor both of these agents. Introductions of other exotic pests may have similar disastrous consequences.

Recently the Southwestern Region of the U.S. Forest Service adopted a new resource management philosophy (USDA Forest Service 1992). This change reflects a desire to take a more holistic approach to management of forests in the Southwest, one that is ecologically based. The focus of this new strategy will be on desired future conditions of the land and its human communities at multiple scales, always striving to maintain a balance between sustaining the resource, lifestyle or social goals, and economic goals. This new strategy emphasizes sustainability, multi-resource management, integrated inventories and analytical tools, and (wherever possible) ecosystem management over single species management. Whenever possible, resource management will "mimic the intensity, frequency, and area extent of naturally occurring disturbance events in an effort to maintain biological diversity and keep disturbance events within the ecosystem's absorbing capability or resiliency" (USDA Forest Service 1992).

Flags which portend catastrophic events that may affect ecosystem resiliency include: excessive uniformity in vegetational composition and structure, large proportion of landscape in high risk conditions for disturbance events, declining species diversity, and habitat diversity loss. Avoiding such catastrophic events will require the use of forest health restoration tools, such as, mimicking natural disturbance regimes through prescribed fire or restorative thinning, maintaining genetic diversity, risk rating to maintain smaller proportions of landscape in high risk for disturbance events, and maintaining disturbance agents and regimes within a normal range of variability.

The long term success of these new strategies will depend, at least in part, upon how well we understand, and incorporate into our management, the effects of insects and diseases on the landscape now and into the future. This will require new knowledge and new analytical tools. New knowledge will be required about insects and diseases, how they affect and have affected ecosystems and ecosystem processes, and how our new management strategies will affect insects and diseases and in turn the landscape. In order to assess implications for ecosystem management we will need to understand these effects at different temporal and spatial scales.

Conclusions

Forest insects and diseases have and will continue to be dominant agents of change in many forest ecosystems. A better understanding of how these agents affect ecosystem functions, processes and linkages is vital to the success of long term ecosystem management.

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Aligning land management objectives with ecological processes in fire-dependent forests

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Abstract. In the years since 1910, wildfire losses of over 1 million acres have only occurred six times. Three of those years have occurred since 1987. In the past decade, fire managers have often been confronted with costly, damaging fires that exceeded suppression efforts. Ironically, many of the most severe wildfires of the past decade have occurred in long-needle pine forests that, a century or more ago, commonly exhibited relatively benign fire behavior. In these forests today, fires are generally more expensive, more dangerous, and more destructive than they were 100 years ago.

During the past century, in the absence of periodic low-intensity burning, these short interval fire-adapted forests have undergone relatively rapid changes in species composition and structure which, in turn, have predisposed them to high-intensity stand replacement wildfire.

In these forests, land management direction that emphasizes basal area growth for wood fiber production, high cover : forage ratios for wildlife habitat, full retention for visual quality, or other objectives that favor late seral stand conditions may be incompatible with the system's ecological dynamics.

Protecting fire-adapted ecosystems cost-effectively and within reasonable limits of risk cannot rely on aggressive fire suppression and fuel treatment alone. Protection will also require that we better align our land management objectives and practices with the ecological processes that fundamentally drive and sustain these systems.

Here in the Southwest, fire has been recognized as an important element across the landscape for a long time. In 1924, Aldo Leopold, while working for the U.S. Forest Service in Albuquerque, remarked on fire's benefits and the consequences of its exclusion in a Journal of Forestry article titled, "Grass, Brush, Timber, and Fire in Southern Arizona." This article's observations challenged early-day Rangers to align their management practices with the ecologies of the land.

Today, fire-dependent ecosystems are at serious risk throughout much of their range. Fire-dependent ecosystems are those where periodic perturbations by fire are essential to the functioning of the system (Heinselman, 1978). In Leopold's time, those sites where fire was most frequent - the desert grasslands - were the first to exhibit ecological dysfunction when fire was excluded. Today, that decline is evidenced in ponderosa pine forests; the next major vegetative type up the moisture/temperature gradient. Ponderosa pine forests range across some 40 million acres in the West.

One-hundred years ago, and for perhaps tens of thousands of years before, low-intensity surface fires swept regularly through ponderosa pine stands. In these systems, the adaptive influence of fire regulated biotic productivity, resilience, and stability in important, but not easily discernible, ways

In the prolonged absence of periodic low-intensity burning, however, changes began to emerge and, at least to some observers, these changes were alarming (Weaver, 1943). Changes in species composition and stand structure began to predispose extensive areas to insect attack, disease outbreak/ and severe wildfire. Through the past decade, in a period of drought, the adverse consequences of fire exclusion have manifest themselves at significant cost.

Not far from here, in 1990, a deadly wildfire named Dude destroyed some 60 homes and killed six firefighters. Although this fire was one of the worst in the area, it shared several common characteristics from

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among many other catastrophic fires in different jurisdictions across the western United States. Fires like Fountain (California 1992); Forty-Niner (California 1989), Aubrey Hall (Oregon 1989), Hangman Hills (Washington 1989), Westberry Trails (South Dakota 1988), Black Tiger (Colorado 1989), Lohman (Idaho 1989), Firestorm (Washington 1991), Foothills (Idaho 1992), Cleveland (California 1992), and Tyee (Washington 1994) were similar:

1. They were among the most costly, most damaging wildfires of the last 10 years.
2. They all threatened homes; most destroyed homes. Some resulted in fatalities; any of them could have.
3. Each cost $250,000 to over $1 million dollars per day to control.
4. They each occurred in ponderosa pine-dominated forests.

Obscured by the ferocity of fires that we typically see in the ponderosa pine type today, is the ironic fact that, 100 years ago, fire behavior in these forests was generally benign. At the turn of the century, fires in ponderosa pine forests maintained the stand. In these forests, today, fires more commonly replace the stand; they are generally much more destructive and much more dangerous than they were 100 years ago.

Through this past drought cycle, we have learned that fires burning in late seral ponderosa pine stands can easily exceed our capabilities to control them. Much discussion is centered on reducing fuel loadings in these areas and that is certainly needed. However, in a more proactive, preventative sense, we also need to re-think our resource expectations in the ponderosa pine type and re-examine those objectives that, in time, predispose this forest to catastrophe. Perhaps too often, we are attempting to maximize various single-resource outputs at the expense of these forest's long-term ecological integrity.

Air quality concerns and the risks involved in using prescribed fire impede our ability to sustain ponderosa pine ecosystems. However, our expectations for timber, wildlife, or visual quality often further exacerbate the problem by emphasizing management practices that are contrary to the dynamics of the system.

Ignoring the importance of function, we often fail to include the ecologies of a system as a factor in managing the resource. In fire-dependent ecosystems, we have often excluded fire and transposed management practices better suited to cooler, wetter sites. The results, now manifest in a period of drought, have been catastrophic.

In ponderosa pine forests before the turn of the century, periodic fires maintained 30-50 trees per acre with sparse understories. Those same stands today often support 300-500 trees per acre, or more (Neueuschwander, 1995; Covington and Moore, 1992; Habeck, 1984). In fire-dependent ponderosa pine forests, when we manage for late seral stand conditions, we are inviting wildfires like Dude, Hangman Hills, and Black Tiger. In these forests, when we manage for:

1. Timber production and maximize basal area growth, or
2. Big-game habitat and maximize cover : forage ratios, or
3. Rare and endangered species and maximize crown density, or
4. Visual screening and maximize understory retention,

we are managing for late seral stand conditions that are contrary to the dynamics of the system. These resource objectives, in these stands, are not realistic over the long-term. From the resource managers perspective, they should be seen as unfeasible. From the firefighters standpoint, they should be seen as dangerous.

In each of the examples cited above, the resource objective requires that we provide full fire protection, curtail silvicultural entries, exclude periodic low-intensity fire, and attempt to hold the stand in stasis. In these situations, we rarely recognize that, in protecting the resource, we are, in fact, placing the forest at risk. We are managing for a high-hazard stand that, over time through episodic drought cycles, has little probability of meeting the resource objective or, in a larger context, ensuring sustainability of the stand.

Some argue that aggressive wildfire suppression policies are to blame for the condition of many fire-dependent forests and that, in restoring them, we should relax fire suppression efforts. Several years ago, when the effects of drought were far less prevalent, it wasn't uncommon to see fires that were "doing nothing but good." More recently, though, under drought conditions, that observation is rarely expressed. The ecological effects of fire are always tied to a specific intensity and duration of fire. In ponderosa pine forests, periodic low-intensity, short-duration surface fires maintain the stand. Other, more intense kinds of fire do not.

Wildfires in ponderosa pine forests are more costly, more destructive, and more dangerous than they should be. We are not going to sustain or consistently win the fire suppression fight in these fire-dependent ecosystems by relying on additional protection capabilities alone. In order to sustain these stands and protect the resources that they provide - and do it safely and economically restoration efforts are going to have to better align resource objectives with the ecological processes that drive these systems.

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Using ecological relationships of wildlife as templates for restoring southwestern forests

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Abstract. We demonstrate approaches to developing conservation strategies for two threatened, endangered, and sensitive predators, the northern goshawk (*Accipiter gentilis*) and the Mexican spotted owl (*Strix occidentalis lucida*), which incorporate knowledge of their habitats and those used by their major prey species. For each predator, a composite of prey habitats is constructed and then synthesized with the predator habitat. This synthesis results in a set of habitats for the predator's food web. Habitats are then projected onto a spatio-temporal scale by referencing them to historic forest conditions. These reference conditions include the composition, structure, and landscape patterns of relevant forest types that existed before intensive management. We focus on forest types because each contains plants and animals that are adapted to local environments; for example, different plant species compositions, structures, patterns, and natural disturbances.

Projection assures that the desired habitats and their mixes are attainable and sustainable; that is, that the desired conditions are within the biophysical capabilities of the vegetation comprising the forest type and that they are manageable. We show how the desired forest conditions, the templates, identified in this approach for Southwest ponderosa pine forests are similar to the natural species composition, structure, and landscape pattern in those forests.

Introduction

Species composition, structure, and landscape pattern of contemporary ponderosa pine (*Pinus ponderosa*) and mixed-conifer forests in the Southwest differ substantially from conditions that existed before European settlement primarily because of grazing, fire management, and logging. While some plant and animal species have benefited from these changes, populations of other species have seriously declined. Because changes have led to increased susceptibility to stand-replacing fires and increased losses to insects and disease, the persistence of the forests and the habitats they contain is of concern (Barrett 1988, Arno and Brown 1991, Reynolds et al. 1992, Wickman 1992, Mutch et al. 1993, USDI 1995).

In response, recent management direction and recommendations emphasize treatments that are consistent with natural ecological processes (Overbay 1992, Reynolds et al. 1992, Risbrudt 1992, USDI 1995). The intent is to maintain a species composition, structure, and landscape pattern similar to that of presettlement forests. However, it is not always clear that managing forests toward natural conditions will conserve populations of species native to a forest type, and this uncertainty is heightened where the habitats of threatened, endangered, and sensitive (TES) species are involved.

We demonstrate approaches to developing conservation strategies for two TES predators, the northern goshawk (*Accipiter gentilis*) and Mexican spotted owl (*Strix occidentalis lucida*), which incorporate knowledge of their habitats and those used by their major prey species. For each predator, a composite of all prey habitats is constructed and then synthesized with the predator habitat. These habitats are then projected at onto a spatio-temporal scale by referencing them against historic forest conditions. Reference conditions

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include the composition, structure, and landscape patterns of relevant forest types before intensive management. This projection is critical for verifying that habitats and their desired mixes are attainable and sustainable; that is, that the desired conditions are within the biophysical capabilities of the vegetation comprising the forest type. Our approach establishes a set of desired forest conditions that vary by forest type because each forest type contains plants and animals that are adapted to local species compositions, structures, landscape patterns, and natural disturbances. This process gives us confidence that species will be conserved.

Habitat provides the life needs of a species; a suitable microclimate, nest sites, food, escape cover, water, and mates. Thus, intrinsic and extrinsic relationships exist between a species and the physical and biological environment where it is found. These relationships partly underlay species evolution (Block and Brennen 1994). Native plants and animals of Southwestern forests were adapted to environmental conditions that existed before the beginning of intensive forest management. Management prescriptions that restore the composition, structure, and spatial patterns of these forests should improve the habitats of native species.

Strategies

Conservation strategies should consider all of a species' requisite resources (nesting and foraging habitats, food) and ecological relationships (competition, predators, diseases) that might limit populations during any part their life histories. Furthermore, because habitats change through succession and natural and anthropogenic disturbances, conservation strategies should address the long-term sustainability of a species' habitat and the habitats of species in its food web. To sustain habitats and food resources, one must understand the habitat relationship of predator and prey and the dynamic physical, biological, and ecological processes that affect the sustainability of their forest habitats at large spatio-temporal scales. Our objective was to design a landscape of sustaining habitats so that long-term goshawk and spotted owl viability was assured.

The first step in developing our strategies was to review the life histories of goshawks and spotted owls including their resource-use, population biology, and habitat ecology. Specifically, we:

1. Reviewed how their populations might be limited by habitat, food, predators, competitors, and diseases;
2. Examined how environmental conditions, primarily vegetation, affect the distribution and abundance of their habitats and influence the types and intensities of their intra- and interspecific relationships;
3. Identified important prey for the goshawk and spotted owl within a forest type;
4. Reviewed current knowledge on the key habitats and foods of their prey;
5. Synthesized this information into qualitative models based on the desired abundance and spatial distribution of habitats of both predators and prey species; and
6. Overlayed information on the vegetation composition, structure, and landscape patterns of a forest type before intensive forest management to give these models a spatio-temporal context.

Incorporating the ecology of a forest, including its patterns of growth and development, allowed us to design a shifting mosaic of predator and prey habitats that simulated the natural shifting mosaic. Our rationale is that forests before Euro-American settlement were relatively stable in ecological time (Hansson 1977, Forman and Godran 1981, Romme and Knight 1982, Delcourt and Delcourt 1983). This stability allowed native plants and animals to adapt to local environments and to one another (Linhart 1989, Linhart et al. 1989). Managing these forests within their natural ranges of variation and ecological limits by using the natural compositions and abundances of fauna and flora and the processes that result in patterns of their assembly, is the best model for sustaining these predators and their food webs.

The shifting mosaic of habitats is achieved by matching, as closely as possible, the desired interspersion of habitats with natural forest patch dynamics. Patch dynamics are the interacting consequences of the life span of trees, stand development characteristics, successional processes and patterns, and natural disturbances that vary in space and time in a forest type. The planning horizons for our conservation strategies are, therefore, centuries.

Choosing the desired landscape

Conservation strategies must identify appropriate spatial scales for providing and assessing changes in the amount, quality, and intermixing of habitats to maximize probabilities for population viability of the target species. The appropriate scale depends on management objectives and can be site-specific or encompass the entire geographic range of the species. For example, if the objective is to maintain only known pairs of goshawks, then the scale should be based on the nesting home range, roughly 4,000 to 10,000 ac

(Eng and Gullion 1962, Reynolds 1983, Kennedy 1990, Hargis et al. 1994, Bright-Smith and Mannan 1994). However, if the objective is to provide for growth of a population by providing favorable habitats for dispersing goshawks, then the appropriate scale may be an entire national forest or ecoregion. Implementing a large scale strategy also allows for the expansion of a predator's home range when prey population are low during winter and lessens the need to determine the "correct" number, spatial configuration, and dispersal corridors between home ranges.

The Mexican spotted owl in pine-oak and mixed-conifer forests of the Southwest requires specific forest structures. Nest-stand structure typically includes large trees, closed-canopies, multiple-height strata, and log and snag decay (Zhou 1994, Ganey and Dick 1995, Seamans and Gutierrez 1995). Foraging habitats are more diverse; the consequence of foraging for prey species that have different habitat requirements (Reynolds et al. 1992, Ganey and Dick 1995). Foraging habitat includes multiple vegetation types and different seral stages (age classes) of those types. The size, shape, juxtaposition, and interspersion of habitat patches are as important to prey populations and to the foraging behavior of the goshawk or owl as the within-patch characteristics. Furthermore, landscape considerations must allow for both short- and long-range dispersal. Short-range dispersal occurs within contiguous landscapes of connected or nearly connected habitats. On the other hand, long-range dispersal occurs between isolated patches of habitat, such as in the Sky Island region of southeastern Arizona where the desired landscape may include stepping stone habitats that facilitate movement between these mountain islands. Conservation strategies must include nonbreeding and dispersal habitats because failure to do so may lead to decreased gene flow and genetic variation, and possibly to local extirpations.

Identifying pre-settlement forest conditions

Understanding the patterns and ecological processes that occurred in Southwestern ponderosa pine forests before the introduction of intensive forestry was central to our approach to developing conservation strategies. Sources of information concerning historical conditions include historical accounts and photographs, tree rings, fire scars, packrat middens, areas that have had minimal human disturbance, dendrochronology, palynology, and paleoecological and forest restoration studies (Kaufmann et al. 1993). Although numerous publications and reports describe the natural conditions of many North American forest types, the breadth and accuracy of the information is highly variable. The following is an abbreviated list of descriptions of the natural composition, structure, and landscape pattern of a variety of forest types: Cooper (1960, 1961), Pearson (1950), Billings (1969), Bormann and Likens (1979), Dieterich (1983), Critchfield (1985), Barbour (1988), Elliott-Fisk (1988), Franklin (1988), Greller (1988), Peet (1988), Laudenslayer et al. (1989), Betancourt et al. (1990), Covington and Moore (1992), McKelevy and Johnson 1992, Grissino-Meyer et al. (1994).

Limitations of pre-settlement data. Knowledge of presettlement forest conditions is largely based on correlative information. Conclusions of studies based on this information for ponderosa pine consistently describe presettlement forests as open, park-like areas composed of large trees that were strongly aggregated into groups. While much evidence supports this generalization, we are concerned with the degree that the results from these studies have been extrapolated. For example, Covington and Moore's (1992) study on the Bar-M canyon watershed of Arizona were on shallow slopes (0-15 percent) and included only one major soil type. We suspect that edaphic differences, topographic influences (especially aspect), and different fire regimes on steeper slopes (>15 percent) would have resulted in different densities, size-class distributions, and species compositions of presettlement trees than on shallow slopes. Many studies of presettlement forests provide few measures of dispersion, or information on distributional properties of the parameters estimated. Whether or not sample sizes in some studies were sufficient for unbiased estimates of the mean number of trees cannot be evaluated without information on sampling variances; sample sizes in most studies were small. While previous work has heuristic value in verifying certain forest conditions, we caution about the extrapolation of these results in complex landscapes.

Northern goshawk

Ponderosa pine in the Southwest (about 2.8 million ac) occurs between the dry pinyon-juniper or oak woodlands and the cool, moist mixed-conifer forests at higher elevations, between 6,500 and 8,500 feet elevation. Within the ponderosa pine zone, the tree occurs either in pure stands where it typically is the climax species, or in mixed-species stands where it is often seral. At lower elevations it frequently mixes with pinyon pine, several species of juniper, and oak; at higher elevations it mixes with Douglas-fir, white fir, blue spruce, southwestern white pine, and quaking aspen (Barrett et al. 1980). Frequent (every 2-7 years) low-intensity ground fires maintained open, park-like forests visually dominated by large yellow-barked pines that occurred in small groups (< 0.3 ac) of trees(Covington

and Moore 1992). Adjacent groups of trees typically differed in ages (Weaver 1951, Cooper 1961, Dieterich 1983, Swetnam and Dieterich 1985). Intensive livestock grazing since the late 19th century, and fire suppression and timber harvests since the early 1900s, have greatly altered the species composition of understory and overstory vegetation, soil characteristics, tree densities, fire fuel loadings, tree vigor, and wildlife habitat (Cooper 1960, Faulk 1970, Covington and Sackett 1984, White 1985, Covington and Moore 1992, Reynolds et al. 1992).

Prey and habitats. The following is a summary of the current knowledge of goshawk and prey habitats historical composition, structure, pattern, and disturbance in Southwest ponderosa pine forests. The complete conservation strategy for the goshawk in this forest type is in Reynolds et al. (1992). Fourteen prey species (or groups of similar species), based on their numerical and biomass contribution to diets, were selected as important prey (Table 1). The majority of these prey reside on the ground or in the lower portions of the tree canopy. All of the mammals, and many of the avian prey, feed on seeds, berries, or the foliage of herbaceous and shrubby plants that occur in the forest understory or in small forest openings. Many feed on pollen and seeds of staminate and ovulate cones of conifers. Tree squirrels, for example, climb trees for cones, while chipmunks and ground squirrels scavenge cones or seeds from the ground or steal cones from caches of others. Other prey species eat arthropods that feed on understory plants or on trees, or that nest in downed logs or in the soil. Mycorrhizal soil fungi are important foods for many of the mammalian prey.

Snags (standing dead trees) provide critical resources for many birds, mammals, invertebrates, and plants. Among goshawk prey, all woodpeckers use snags for feeding, nesting, or both. Several other species of birds use snags for perches. Four mammals use snags with cavities for nesting and for caching cones.

Large, downed logs provide cover, feeding, and nest sites for a variety of vertebrates. Among goshawk prey, downed logs are important feeding sites for woodpeckers and as denning sites for chipmunks, mantled ground squirrels, and cottontail rabbits. Downed logs are also used by blue grouse during courtship.

The character, amount, and distribution of woody debris (material >3 in. and <12 in. in diameter) may affect the abundance of goshawk prey (Dimock 1974). Woody debris provides important denning sites for cottontail rabbits, cover for rabbits, chipmunks, and ground squirrels, and feeding sites for several woodpecker species.

Large trees (>18 in. in diameter) provide critical nesting, denning, feeding, and roosting sites for such goshawk prey as tassel-eared squirrels, large woodpeckers, and blue grouse. Large trees also are good cone producers, providing seed for many prey species. Large trees are the only source for large snags and downed logs, both important to all but one member of the suite of prey. Large trees also provide hunting perches and nest trees for goshawks.

Openings with their associated grassy, herbaceous, or shrubby vegetation, provide important food and cover for a number of goshawk prey. Three species require openings; blue grouse for nesting and brood-rearing, and band-tailed pigeons and mourning doves for feeding. Because pigeons and doves typically travel long distances to feed, large openings may not be necessary for them. Small to medium openings (<4 ac) benefit blue grouse, chipmunks, and mantled ground squirrels, and minimize the effects of larger openings on the more interior forest species.

Herbaceous and shrubby understories provide important foods (seeds and berries) and cover for many of the selected goshawk prey. Well developed understories occur in forests with canopies sufficiently open to allow adequate light to reach the forest floor. Herbaceous and shrubby understories are critical foods and cover for robins, doves, pigeons, grouse, chipmunks, rabbits, ground squirrels.

Interspersion is a measure of the degree of intermixing of vegetation structural stages (VSS). VSS describe forests on the basis of tree-size classes from seedlings to old trees: VSS 1 is open area dominated by grasses, forbs, and shrubs; VSS 2 is dominated by seedlings and saplings; VSS 3 is young forests; VSS 4 is mid-aged forests; VSS 5 is mature forests; and VSS 6 is old forests. Several prey require relatively high structural stage interspersion levels, others are affected little or not at all by interspersion. Tassel-eared squirrels, blue grouse, and cottontails need relatively high interspersion levels, while chipmunks are affected little by interspersion. Although some prey occur in each VSS, the mid-aged, mature, and old age-classes (VSS 4-6) are significant to most (11 of 14) important prey (Reynolds et al. 1992). Some species (American robin, mourning

Table 1. Suite of important northern goshawk prey in the Southwest (from Reynolds et al. 1992).

Birds	Mammals
American robin	Chipmunks (*Tamius* spp.)
Band-tailed pigeon	Cottontail (*Sylvilagus* spp.)
Blue grouse	Mantled ground squirrel
Hairy woodpecker	Red squirrel
Mourning dove	Tassel-eared squirrel
Northern flicker	
Red-naped sapsucker	
Steller's jay	
Williamson's sapsucker	

dove) are generalists and occur in most VSSs, while others, such as Williamson's sapsucker, occur in a limited number of structural stages. Blue grouse need older forests (VSS 4, 5, and 6) interspersed with openings (VSS 1), and tassel-eared squirrels need a mix of mid-aged and mature or old forests.

Many of the mammalian prey species depend on fungi during summer and fall; the physiological condition in which tree squirrels begin the winter depends on the amount of fungi eaten (Maser et al. 1978). In ponderosa pine forests, the best fungi-producers are mid-aged VSS with high canopy cover and shade to protect soil moisture (States 1985, States et al. 1988, Uphoff 1990).

Mexican spotted owl

The Mexican spotted owl was listed as a threatened species in 1993 under the Endangered Species Act (1973) and a team was formed to develop a recovery plan (USDI 1995). Summarized below are features of the recovery plan that characterize habitat needs for the owl and its prey.

Prey and habitats. Seven groups of vertebrates, including more than 16 species, are important prey for owls based on comprising >10 percent of the diet by either frequency or biomass (Table 2). Although the owl's diet varies geographically, woodrats (*Neotoma* spp.) and peromyscid mice (*Peromyscus* spp.) usually dominate the diet. Even so, the species taken as prey represent a diverse array of habitats and ecological niches. Considered as a group, they demonstrate the importance of managing for a landscape mosaic consisting of habitat patches in various conditions and seral stages.

Although the owl is found predominantly in mixed-conifer forests, some reside in ponderosa pine-Gambel oak forests of northern Arizona. By biomass, the most common prey in these forests are woodrats (*N. mexicana* and *N. albigula*), peromyscid mice (*P. maniculatus* and *P. boylii*), cottontail (*Sylvilagus nuttalli*), and various sciurids (*Spermophilus lateralis*, *S. variegatus*, *Tamias dorsalis*, *T. cinereicollus*). Lesser components of the diet include various bird species, pocket gophers (*Thomomys* spp), Mexican voles (*Microtus mexicanus*), bats, and arthropods.

Habitat requirements for each of these species are unique. The deer mouse inhabits most areas, whereas the brush mouse and Mexican woodrat favor areas with rocky outcrops, high log volume, and a shrub understory. Such conditions occur frequently, but not exclusively, on poor timber producing sites. Cottontail, pocket gophers, and voles use more open areas where sparse canopy cover allows for a well-developed herbaceous understory. These areas typically occur within stands, along drainages and canyon bottoms, or in meadows adjacent to forests. Curiously, voles contributed little to the total diet of owls in the pine-oak forest, although they are a major diet component in other parts of the owl's range (Ward and Block 1995). The low number of voles

Table 2. Prey comprising >10% of relative frequency (X) or biomass (O) in the diet of Mexican spotted owls (from USDI 1995).

Prey group	Colorado Plateau	Southern Rocky Mountains Colorado	Southern Rocky Mountains New Mexico	Upper Gila Mts.	Basin & Range West	Basin & Range East	Sierra Madre Occidental Norte
Bats	X			X	X		
Rabbits	O	O	O	O	O	O	O
Pocket gophers				O			O
Peromycid mice							
Deer mouse	X O	X O	X	X O	X O	X O	X O
Brush mouse	X O	X O	X	X O	X O	X O	
Canyon mouse	? ?						
Woodrats							
Mexican woodrat	X O	X O	X O	X O	X O	X O	X O
Bushy-tailed woodrat	X O	X O	X O				
Desert woodrat	X O						
White-throated woodrat		X O	X O		X O		
Voles							
Mexican vole		X O		X O	X O		
Mountain vole		X O	X O				
Meadow vole		X O	X O				
Long-tailed vole		X O		X O	X O		
Birds			X O	X	X O	O	
Arthropods	X	-[a]	X	X	X X	X	

[a] *Undetermined.*

in the diet was corroborated by low numbers captured during live-trap sampling (Block and Ganey, unpubl. data). Possibly the unnaturally dense conditions of these forests (Covington and Moore 1992) have suppressed the growth of grasses and forbs decreasing habitat quality and quantity for species like voles, gophers, and rabbits.

Providing appropriate habitat conditions for prey species is critical for buffering predator populations against natural population declines in any prey species. Given the variety of prey taken and differences among prey species in habitat requirements, the desired landscape should be a mosaic consisting of vegetation in different conditions and seral stages. These prey species are adapted to natural conditions in Southwestern forests; thus, returning these forests to more natural conditions will help ensure an adequate prey base for the Mexican spotted owl.

Templates: Desired forest conditions

Northern goshawk. Our analysis showed that goshawk foraging habitat consisted of large trees and relatively open understories. Large trees have large limbs that are used as hunting perches, and relatively open understories provide flight room and opportunities for detecting and capturing prey. Habitat for goshawk prey in Southwestern ponderosa pine forests included small, scattered openings and a high interspersion of age classes to provide a diversity of habitats and foods. On average, canopy cover was relatively open, allowing sunlight and moisture to reach the forest floor so that grassy, herbaceous, or shrubby understories could develop. However, groups of trees within patches of VSS 5 & 6 had interlocking crowns and higher (60-80 percent) canopy cover. Interlocking crowns provided travel routes for squirrels (Patton 1984) and other prey, and shading to protect soil moisture and fungi populations (States 1985). Large tree components (live trees, snags, and downed logs) were abundant and scattered throughout the landscape. Large live trees provided many unique hiding, feeding, denning, and nesting sites used during some part of the annual cycle of all important goshawk prey species. Goshawk habitat in ponderosa pine should have abundant prey when at least 60 percent of the landscape is in the three older age classes (VSS 4, 5, and 6).

The desired habitats of goshawks and their prey in ponderosa pine consisted of a mosaic of highly interspersed, small patches of different structural stages (grass/forb/shrub stage to old forest stage). This mosaic of forest patches and within-patch structure closely resembled the species composition, structure, and landscape pattern in historic Southwestern ponderosa pine forests. Mature and old trees in these pine forests were strongly aggregated into groups consisting of 3 to 44 trees and occupied from 0.5 to 0.7 ac (Cooper 1961, White 1985). Groups were typically separated by variably-sized openings. Although openings appeared unstocked, the roots of trees in the groups extended 50 feet or more into the openings (Pearson 1950). Tree reproduction (seedlings, saplings, poles) either occurred within a group, often when a mature tree fell (Covington and Moore 1992), or in openings between groups (Pearson 1950, Cooper 1961). Seedlings were established in a year when fire exposed the mineral soil seed bed and when there was high seed production and rainfall (Cooper 1960, 1961). The historical ponderosa pine forest landscape was composed of an all-aged forest (coarse scale) made up of small, either even-aged or multi-aged patches of trees (fine scale).

Interlocking tree crowns, large limbs in and below crowns, and extensive shading are important within-patch characteristics for goshawks and their prey. Yet trees in these patches should be vigorous enough to produce abundant symbiotic mycorrhizal fungi. Trees within groups of mature ponderosa pine display the characteristics of open-grown trees and trees grown in dense stands. Interior trees are under greater competition for soil moisture, nutrients, and light, whereas trees on group edges have more sunlight and openings to spread their roots (Pearson 1950). While interior trees have narrow and short crowns, trees on the edge have one-sided but long crowns. Further, the crowded conditions within groups result in an intermingling of adjacent trees limbs. The long crowns of the outside trees provide shading and maintain higher soil moisture within the group, which are conditions ideal for mycorrhizal fungi production (States 1985, States et al. 1988, Uphoff 1990).

Large diameter snags were historically abundant and well dispersed in Southwestern ponderosa forests because of the abundance and susceptibility of old trees to mortality factors such as wind, lightning, fire, mistletoe, fungal diseases, insects and other herbivores (Pearson 1950, Cooper 1961). Although frequent ground fires kept the forest floor relatively clear of woody debris, portions of large diameter logs often survived fires. These large trees and snags provided a continuing source of large-log habitats for goshawk prey.

The goshawk conservation strategy identified the proportions of managed landscapes that could be maintained in specified age classes so that the availability of those age classes would remain somewhat constant through time. For the goshawk, this objective was met when a landscape had 20 percent in each of the older four age classes (VSS 3, 4, 5, 6) and 10 percent each in VSS 1 and 2. The proportions that can be maintained in the different age classes depends on the number of years required for seedling establishment, growth rate of trees (partly dependent on the intensity of management), and tree longevity.

Mexican spotted owl. Similar to the desired conditions for goshawks, the forested landscape for Mexican spotted owls and their prey in ponderosa pine-Gambel oak and mixed-conifer forests should emphasize a mosaic of different seral stages. Although the interspersion and juxtaposition of patches was unknown, a key consideration was the distribution of seral stages as they relate to topography, specifically slope and aspect. Presently, most owl nest and roost sites in this forest type are on steep slopes associated with cinder cones and drainages. Whether or not presettlement forest conditions on these slopes differed from those on flatter slopes is unknown. However, because of edaphic and microclimatic differences, and possibly differences in fire regimes, assuming that presettlement forest structure varied with slope and aspect is reasonable. Research is needed to describe the variations in presettlement forest structure as influenced by environmental differences.

The limiting factor for owls is adequate nest and roost habitat. Typically, such habitat exhibits high tree basal area, large trees, and a large oak component. Most nests in this forest type are in cavities of large Gambel oaks. Gambel oak are shade intolerant, thus they require an open canopy structure to become established and achieve maximum growth potential. The size of nest stands is unknown but is currently under investigation (R. Gutiérrez, personal communication with W. Block). However, the Forest Service stand database indicates that 7 to 10 percent of pine-oak forests meet nest stand characteristics (USDI 1995). Whether or not stands exhibiting nest characteristics existed before European settlement and, if so, what proportion of the landscape contained these conditions, is unknown, but should be researched. The remainder of the landscape should include stands of various seral stages to meet the habitat requirements of the prey the owl requires. The distribution of seral stages and mosaic patterns required by the goshawk in pine forests appear compatible with spotted owl needs. We assume that species in both the owl and goshawk food webs are adapted to natural conditions; therefore, even partial restoration of those conditions will improve their habitats.

Implementation

A conservation strategy is not complete without area-specific guidelines for its implemention. Attainment of the desired conditions requires an understanding of the relationship between existing forest conditions, the desired conditions, and the landbase capability. The choice and intensity of management should be based on the direction and the distance the existing forest conditions must move to reach the desired conditions. Both Reynolds et al. (1992) and the USDI (1995) recognized that the capability of a landbase to produce certain forest conditions varies across sites and that this variation is associated with elevation, slope, aspect, soil, and moisture and nutrient availability. During implementation, we suggest the best reference conditions for sustaining habitats on sites are the historical species compositions, densities, and patterns on those sites.

Present forest conditions are moving away from historic conditions; many of the recommendations in our conservation strategies reverse that direction. Specific management prescriptions for Southwestern ponderosa pine were designed to improve predator and prey habitats by:

1. Increasing the abundance of old trees and forests, large snags, and large, downed logs;
2. Restoring the grouped nature of trees and the interspersion of small patches of different age classes;
3. Restoring the habitats and foods provided by a well-developed grass, forb, and shrub layer in understories; and
4. Protecting habitats from catastrophic loss from fire and insect epidemics by reducing fuel ladders and tree overstocking.

Some recommended tools to attain these goals are small group-selection (uneven-aged management), retention of large snags and logs, protection of the organic surface layer of soils during management activities, lengthening the harvest cycle, thinning from below, and the reintroduction of ground fires (Reynolds et al. 1992, USDI 1995).

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Implementing adaptive ecosystem restoration in western long-needled pine forests

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Abstract. This paper discusses the restoration of western long-needled pine ecosystems in general with a focus on developing adaptive ecosystem management projects which will simultaneously restore these ecosystems and advance basic understanding of how these systems operate. The paper begins with some background remarks on ecosystem restoration. Next comes a brief overview of the evolutionary context of long-needled pine forests of western North America. Then I present a broad overview of human caused changes in the structure and function of long-needled pine forests. Next I illustrate these changes for the ponderosa pine forests around Flagstaff, Arizona. Finally, I close with a call for interagency cooperation to implement adaptive ecosystem restoration of western long-needled pine ecosystems at an operational scale.

Introduction

A fundamental postulate of ecosystem management is that restoring and managing ecosystems consistent with conditions present during their evolutionary history is the most effective strategy for preserving diversity, maintaining endangered species, and avoiding catastrophic disruption of ecosystem functioning (Society for Ecological Restoration, 1993). Ecological restoration rests on the premise that the entire ecosystem will function best under the conditions to which its component organisms have become adapted over evolutionary time. Restoration does not mean that the ecosystem can be returned completely to the presettlement era nor does it imply a rigid, uniform prescription for management of every acre of forest land. In fact, on most of the land, restoration will be used to maximize compatibility between both natural processes and structures and human habitat requirements.

For western long-needled pine forests, ecosystem restoration implies that dense patches of postsettlement trees should be thinned to promote tree and grass growth and vigor; that native grasses, shrubs, and wildflowers be encouraged to provide forage for wild and domestic animals as well as enriching the soil and holding moisture; that a range of tree ages, especially of the oldest trees, be maintained to ensure habitat and genetic diversity; that heavy, unnatural postsettlement fuels be treated and then that prescribed fire be reintroduced on regular intervals to carry out its natural role. Within this broad-based approach, there is room for emphasis on specific goals for timber, range, water, or wildlife and game production, recreational opportunities, and human homesites. As the field trips and talks at this conference have shown we are a long way from healthy ecosystems such as these. Today all resources and ecological processes are suffering from current forest conditions.

An evolutionary context

Ponderosa pine is the most widespread member of an ecologically similar group of long-needled pine in the section ponderosae. Principal members of this group are Arizona pine (formerly classified as a five-needled subspecies of ponderosa pine), Durango pine, Apache pine, and Jeffrey pine. These species share the morphological characteristics of having thick bark, protected buds, prolific seed production, longevity, and abundant and highly flammable litterfall, all of which are considered to be adaptations to frequent, low-intensity surface fires. They are analogous to the red pine forests of the Great Lakes region of North America and long-leaf pine of the southeastern United States.

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The earliest paleoecological record of this group of yellow pines comes from 50 million-year-old macrofossils found in British Columbia. Ponderosa pine macrofossils dating from 26.5 million years ago have been found near Creed, Colorado. Throughout the late Pliocene and the Quaternary ice ages species of ponderosa pine/bunchgrass ecosystems have migrated up and down in elevation and latitude tracking favorable climatic conditions. At various points in time, ponderosa pine/bunchgrass communities were much more prevalent, most notably during the Pliocene (2-5 million years ago) when these ecosystems occupied 200-300 million acres of North America and provided extensive habitat for the modern biota of today as well as other some species now extinct, including some species of the prehistoric megafauna such as mammoths, ground sloths, and saber toothed tigers. Clearly, ponderosa pine/bunchgrass ecological systems have coevolved with frequent surface fires and open park-like stand conditions for many millions of years.

Since Pliocene times these forests have provided important evolutionary habitat for an exceptionally diverse biota, much of which appears to be adapted to frequent fire. For the past 10-30 thousand years these forests have been vital resources for numerous human cultures, most recently (1850-present) for Euro-American industrialization.

Early human cultures in North American forests supplemented lightning ignitions by using fire as a hunting, gathering, and agricultural tool. Native Americans used fire to extend the range of ponderosa pine parklands into adjacent forest and woodland types (Arno 1985). Soon after Euro-American settlement of the region, a period of intense resource exploitation began during which the ecosystem capital of large old-growth trees and lush herbaceous vegetation generated tremendous wealth for the rapid expansion of the then infant Euro-American economy. However, intensive exploitation exacted its toll on the ecosystem, setting into motion changes which would result in the depauperate conditions we see today.

Postsettlement changes in Southwestern ponderosa pine

Old-growth tree populations and their dependent communities have been, and are continuing to crash precipitously, first from logging and then from competition with irrupting postsettlement tree populations and crownfire. Heavy livestock grazing broke grass fuel continuity and active fire suppression eliminated the presettlement fire regime. In the absence of frequent fires striking changes occurred: tree species less adapted to frequent fire have invaded (at the expense of other plants), and conifer tree biomass, both live and dead, has steadily accumulated, contributing to progressively declining biodiversity, increasing susceptibility to insect and disease epidemics, and supporting a shift from frequent, low intensity surface fires to larger and larger crownfires (Cooper 1960, Covington and Moore 1994b, Swetnam and Baisan 1994).

To a society with high demands for wood products, the increase in tree density at first seemed beneficial to many. However, after 50-70 years of fire exclusion, foresters and ecologists, beginning with Aldo Leopold in the 1930's, began sounding the alarm that fire exclusion in these long-needled pine forests was leading to rapidly accelerating ecological degradation. For example, Harold Weaver in 1943 summarized conditions in eastside Washington ponderosa pine:

> "Dense, even-aged stands of ponderosa-pine reproduction have developed...enormous areas are growing up to dense, even-aged stands of white-fir [sic], Douglas-fir, and incense-cedar [sic] reproduction under the merchantable ponderosa pines...for the past 20 years epidemics of the western pine beetle have killed and are continuing to kill billions of board feet of ponderosa pine worth many millions of dollars. Because of these ecological changes, which are continuing to take place, the fire hazard has increased tremendously. Fires, when they do occur, are exceedingly hot and destructive and are turning extensive areas of forest into brush fields."

Soon other researchers pointed out additional undesirable consequences of fire exclusion in ponderosa pine forests. Studies in Utah (Madany and West 1983, Stein 1987), Montana (Gruell et al. 1982), Idaho (Steele et al. 1986), Washington (Weaver 1943), California (Laudenslayer et al. 1989), and the Southwest (Cooper 1960, Covington and Sackett 1986, Covington and Moore 1994a,b) have shown that increased tree density, fuel loading, and crownfire occurrence are common consequences of fire exclusion throughout the ponderosa pine type.

Various authors (e.g., Arnold 1950, Cooper 1960, Biswell 1972, Weaver 1974, Kilgore 1981, Williams et al. 1993, Covington and Moore 1994a,b) have inferred that associated with these increases in tree density, forest floor depth, and fuel loading in ponderosa pine ecosystems have been:

1. Decreases in soil moisture and nutrient availability;
2. Decreases in net productivity and diversity of herbaceous plants and shrubs;
3. Decreases in tree vigor, especially in the oldest age class of pine;
4. Decreases in animal productivity;
5. Decreases in stream and spring flows;

6. Increases in susceptibility to pine bark beetles; and
7. Increases in fire severity and size.

In sum, the implication is that today's tree densities and fuel loads in ponderosa pine ecosystems are not sustainable. However, with few exceptions these inferences have not been supported by intensive ecosystem management-oriented research.

Public recognition of the severity of these ecological changes has led to considerable debate over implications of various management scenarios (including no action) on ecosystem health and sustainability. Furthermore, researchers, other natural resource professionals, and the lay public are embroiled in an often rancorous debate over what, if anything, should be done. Concerns about overcutting of old-growth trees (or for some factions practically any commodity uses of forestlands) has led some environmental groups and some scientists to argue against any role for mechanical treatments in restoration of ponderosa pine ecosystem health. However, others point to evidence that without mechanical treatment to reduce unnatural fuel loads, the ensuing fires, even under controlled conditions, can kill old-growth trees and other vegetation, and cause such intense soil heating that restoration of natural conditions is retarded, if not precluded, for the foreseeable future (see review by Covington et al. 1994). Aldo Leopold suggested several lines of evidence for a potential synergy between innovative commodity resource uses and restoration and maintenance of ecosystem health:

> "... Leopold set out to define conservation in the following terms: as 'a universal symbiosis with land, economic and aesthetic, public and private;' as 'a protest against destructive land use;' as an effort 'to preserve both utility and beauty;' as 'a positive exercise of skill and insight, not merely a negative exercise of abstinence and caution;' and, finally, as 'a state of harmony between men and land.'" (passage from Callicott 1994).

However, objective scientific data to support such management actions are inadequate.

Systematic field research in combination with synthesis from existing knowledge can help fill this information gap by providing a sound scientific basis for evaluating the consequences of various ecosystem management options and designing adaptive ecosystem management projects. It is a fairly straightforward task to determine the effects of wildfires, prescribed fire, understory thinning, and bark beetle-induced tree mortality on key ecosystem and human resource characteristics of long-needled pine ecosystems. Collection of field data in combination with synthesis of historical data and ecologically-based response functions can be used to examine the relative effects of fire, cutting, and bark beetle infestation treatments on:

1. Tree composition, density, spatial pattern, size, age structure, growth efficiency, and biomass;
2. Fuel composition and structure;
3. Herbaceous and shrub composition and biomass; and
4. Selected wildlife and human habitat values.

Because ecosystem management uses the concept of the range of natural or historical variability as a key reference point (Morgan et al. 1994), a fundamental comparison should be analysis of treatment effects in relation to conditions which prevailed before disruption of the natural fire regime in these forests.

Examples of restoration at work

The impacts of Euro-American settlement in ponderosa pine ecosystems has been devastating to the native biota. The basic chain of events, familiar to us all, consisted of cutting out the old-growth trees, extirpation of predators, introduction and subsequent irruption of livestock populations, and as a consequence, disruption of natural fire regimes.

Dendrochronological analysis of multiple fire scars from the Chimney Spring Interval Burning Study Area 7 miles north of Flagstaff indicates that the average fire interval on this site was 2.3 years before fire regime disruption in 1877 (Dieterich 1980). Beginning in 1877 thousands of head of cattle were introduced into the area. The ensuing overgrazing eliminated the herbaceous fuels which had carried fires across the landscape holding pine populations in check.

With the completion of the transcontinental railroad in 1882, logging began in earnest in the Flagstaff vicinity. Thus began a population crash of the largest and oldest trees which continues to the present. The next event was an irruption of dense forest stands and a crash of herb- and shrub-based food webs. These tree population irruptions are apparent not only at the landscape level but also within stands. Photo series from a restoration study at the Pearson Natural Area north of Flagstaff show open forest conditions in 1909, the seedling population explosion by 1938, and the dense sapling and pole thickets of today which compete with the old-growth trees and provide ladder fuels for fires to reach their crowns (Covington and Moore 1994a). Similar photo series are available from Montana, Oregon, South Dakota, California, and elsewhere. The upshot of this tree population irruption has been the conversion of diverse park-like stands to dense forests. In a very real sense what we have witnessed throughout the western long-needled pine forests is the flip-side of

forest fragmentation: the fragmentation of the once vast herbaceous and shrub vegetation which once served as the surface matrix for these diverse and productive parklike pine forests. From a biodiversity standpoint, these tree population irruptions have caused a tremendous simplification of net primary productivity to the point at which today virtually all is concentrated in trees.

The reconstructed sequence of events for the Bar-M study area 25 miles south of Flagstaff is instructive (Covington and Moore 1994a). Steadily increasing tree density has lead to increasing crown closure, a continuous fuel ladder, heavy forest floor accumulations, and a crash of herbaceous production. Perhaps most devastating has been the irruption of increasingly large and devastating crownfires, which were not part of the evolutionary experience of most of these ecological systems. When you think about it, it would be very difficult to design a more devastating assault on the biodiversity of ponderosa pine/bunchgrass ecosystems. Imagine the outcome of an environmental assessment of fire exclusion conducted in the 1870's if they knew then what we know now.

Conclusion

Although at differing stages, these transformations are ubiquitous throughout the range of ponderosa pine from Canada to Mexico (Covington et al. 1994). In addition to areas in the Southwest, I have visited sites from through out this range from Kamloops, British Columbia, the Black Hills (South Dakota), Colville Indian Reservation in Eastern Washington, the Sierra Nevada of California, the Sierra Madre Occidental of Mexico. In Mexico, some still burn on a 3-10 year interval (Fulé and Covington 1994, Fulé and Covington, this volume).

Setting these ecosystems on a more sustainable path is straightforward. What is needed are interagency cooperators of practitioners and researchers from throughout this range who are interested in implementing ecosystem restoration in an adaptive ecosystem management context (Williams et al. 1993). A team of scientists at NAU is working on one such project with the Arizona Strip District of the Bureau of Land Management (Taylor, this volume). In that project we are initiating cooperative ecosystem management work to answer questions regarding whether existing disturbances can restore ecosystem structure and function, to test hypotheses regarding ecosystem restoration treatments, and to establish demonstration areas to serve as public information and education areas.

In conclusion, it seems clear that ecological restoration of western long-needled pine ecosystems offers unparalleled opportunities for implementing ecosystem management with a win/win outcome. We could restore ecological integrity and improve resource values, paid for at least in part by removal of postsettlement trees. The risks of inaction far outweigh the risks of implementing ecosystem restoration.

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The Gerber Block, a Continuum in Stewardship on Bureau of Land Management Land in Klamath County Oregon

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Abstract. The Gerber Block, a 112,000 acre area managed by the Bureau of Land Management in south central Oregon, has a long history of public use for a variety of commodities and experiences. This, combined with tenured management with a strong commitment to maintenance of sustainable, as well as aesthetic, biological systems has resulted in an area perhaps unique in managed ponderosa pine systems. Forest management over the past 35 years has combined applied silvicultural treatments with individual commitment to preservation of the aesthetic appeal of old growth and a maintenance of diverse stand structure.

Over the past thirty-five years only three foresters and one forestry technician have shared a continuum of stewardship for the management of the forested woodlands and ponderosa pine stands in the Gerber area. These individuals, supported by their line managers and sharing the expertise of other specialists, had freedom to use personnel discretion and deviate from prevalent thinking of the time. They have not been confined by specific manuals and handbook guidance. Certainly a sense of personal pride and stewardship prevailed.

Most of the pine stands have been selectively logged three to four times. An aggressive precommercial thinning program in the sixties and seventies reduced stocking to reasonable levels within site capacity. Although fuel loads are still above acceptable levels in many areas, an active underburning program started in 1980 has reduced the risks associated with continuous fuel loads. Certainly the earlier thinning has complimented this effort. The retention of snags for wildlife use began in the middle sixties. The objective to retain a significant amount of old growth because it "looked nice," maintained the historic appearance of the pine forest and was necessary for partial shade for regeneration on dry sites. Rehabilitation, reconstruction and/or obliteration of roads began in earnest in the sixties.

Today the terminology has changed and objectives for sustained management of ponderosa pine are being revised from the thinking of the sixties and seventies. However, perhaps by accident or simple blind luck, there are those scattered areas where past management practices have left a legacy that allows us insight to the future. Perhaps the Gerber block is just such an area.

Introduction

There are scattered forests and rangelands across the landscape where past management practices left a piece of land in a condition which reflects both forethought in management direction and the dedication of numerous resource management specialists and stakeholders to land stewardship. The "Gerber Block" described in this paper may be just such a place.

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Located 50 miles east of Klamath Falls, Oregon and adjacent to the California border is a 112,000 acre area of public land managed out of the Lakeview District, Klamath Falls Resource Area. Elevations range from 4200 to 5400 feet. The land is relatively flat and rocky. Vegetation includes approximately 46,000 acres of sagebrush and grass, 50,000 acres of western juniper woodland, and 9,000 acres of ponderosa pine. The pine commonly occurs as the single commercial species with only occasional scattered white fir at highest elevations. The pine is commonly found as "stringers" or narrow bands below low rims and adjacent to rocky flats. In structure the stands consist of even-aged clumps in uneven-aged stands.

Early management

I arrived in the area in 1966 as a young forester on his second field assignment. After spending over three years in the forests of central Oregon I was no stranger to open grown stands of ponderosa pine. The last thirty five years has caused enormous changes in the structure and condition of our ponderosa stands as some of you who watched the forest transform over time can attest. At that time regeneration was still moving in earnest into the open stands. Inventories in the 60's showed a shortage of stands in the pole and small sawtimber sizes.

Federal agencies had begun aggressive precommercial thinning programs in many locals. It is my opinion and supported by historical photos and study plots, that we were much closer to historical ranges in 1960 then we are today. Conditions in the Gerber Block were similar to those found in the Ochoco, Deschutes, and Fremont National Forests with which I was familiar. I would be interested to see records on the decrease in density reduction practices over the past thirty years on Federal forest land. I think they would be significant.

The influence of one man, a forester named Bruce Whitmarsh, had a lot to do with what became the management direction for forestry in the area. His tenure was from 1957-61 and 63-69. Except for localized high risk cutting in the 40's and some entries by crews out of the Bend office in the early 50's, many stands had not yet been entered. There had been considerable road building with the first entry, and I use the term loosely, which consisted of scraping a track across the scab rock with little regard to future drainage. These tracks suffered significant erosion and one objective during the second entry was the rehabilitation and stabilization of those roads. In spite of this, the forest was still relatively undisturbed. Keen's Crown Class directions for evaluating ponderosa pine was commonly used and only those trees which were not expected to last at least ten years were cut. Cutting was light probably averaging only 1-3 MBF/acre where high-risk trees were present.

Bruce was elated to have a young impressionable forester replace the "Woods Butcher" as he referred to the individual who had moved in on his territory in his absence between 1961-63. Bruce had done his best to keep a paint gun out of the "Butchers" hands. I spent eight happy years there and turned it over to silviculturist Bill Johnson in 1974. He has been there ever since.

Early stand conditions

Frequent fire scars attest to lightning fires which had historically burned through the pine stands every 8-12 years. When Henry Gannett did his paper on "The Forests of Oregon" in 1902, he reported, "where the forests are largely or mainly of yellow pine in open growth, with very little litter or underbrush, destructive fires have been few and small, although throughout these regions there are few trees which are not marked by fire, without, however, doing them any serious damage."

These fires limited understory woody vegetation and continuous tree reproduction while favoring grasses, park-like clumps of old growth, and scattered groups of reproduction. Direct suppression of fires and change in frequency began with livestock grazing along with fire control by local residents and government agencies. As a result, ground fuels and understory growth began increasing about 1900 along with changes in vertical arrangement of fuels.

The stands we began to work with in the mid sixties reflected the influences described above. We still had numerous clumps of old growth with no understory. It was our intent to open up the old growth enough to get reproduction back but still maintain a strong old growth component. One third to half of the stems were removed on the second entry and included those most suppressed individuals along with a thinning of dense clumps of old growth. Younger clumps were commercially thinned where possible but diameter limits for utilization only went down to 12" DBH with an 8 inch top and one 16 foot log.

Management guidelines for forests

Between the mid sixties and early seventies the following management practices were applied. They have changed little to this day except for more restricted removal of larger size classes.

- We began leaving snags as well as old isolated trees in openings and along the edges of the numerous rocky flats adjacent to timbered areas. This was done primarily for wildlife concerns but also to reduce fuel on the ground were the trees to be felled, and for scenic purposes. Even where limited high-risk marking had occurred, the original stands were virtually intact. Leaving snags at that time left us wide open to peer criticism.
- We stuck our neck out and began requiring precommercial thinning of suitable stands within the sale area boundary as part of the timber sale contract requirements. Purchasers were given the choice of doing the work or depositing the money for us to contract it. Between 1962 and 1974 approximately 2,200 acres were thinned which amounted to in excess of 80% of treatable stands.

- Rehabilitation of roads began with the second entry. Where roads existed and were stable, no use of a blade was permitted. Frequent cross-drains were provided. These were usually in the form of dips. Use of culverts was kept to a minimum thereby reducing maintenance needs. A small rangeland drill was used to seed grass and legumes on disturbed areas. This was far superior to broadcast seeding. Main haul roads were constructed in the early 60's to higher standards and eventually surfaced although, they were still just adequate to serve their intended purpose.
- Dirt pads or concrete fords were used in intermittent drainage crossings and removed following use. Minimum standards were used and some roads were closed completely by pulling the berm and numerous rocks back onto the road surface.
- While marking, special attention was given to limiting damage to residual trees. Slash was only piled when in concentrations and on landings. Trees were limbed full length to get material on the ground where decay would occur. This also avoided excessive concentrations of slash.

Influence on restoration strategy

Several things were happening here which influenced the structure and condition of the forest and made it easier to work toward a pre-suppression condition when fire was introduced.

- There was almost no, except in isolated areas, broad-scale overstory removal as was done on much of the adjacent Fremont National Forest. An old growth component was retained in virtually all cases with the second entry and in most cases with the third entry.
- Precommercial thinning slash had, to a large degree deteriorated on the ground prior to the reintroduction of fire beginning in 1980. While this still left considerable fuel loads, growth rates had increased with the reduced basal areas and many trees would reach merchantable size within the next twenty years for removal with the third entry. Also, the thinned stands were easier to burn.

Rangeland restoration

Some other things were happening in the area which increased continuous fuel across the landscape. The area had a history of early season grazing before livestock moved onto the National Forests. Local ranchers lobbied for active management of the public lands in the area. The Bonanza Grazing District, of which the Gerber Block was a part, was established in 1935. It was the first functional grazing district under the Taylor Grazing Act. Henry Gerber was a member of the first National Advisory Board. At that time the block was divided into allotments and grazing seasons were set.

In the late 1960,s Dr. August Hormay's teachings on rest rotation grazing had an important influence on the area. Wherever possible, allotment management plans were set up. Some of these plans have been in operation for over twenty years. Additional pastures were fenced, and wildlife use was given significant consideration. Parts of the area are important deer winter range and antelope are numerous. In recent years, rocky mountain elk have moved into the area in significant numbers, turkeys have been introduced. Bald and golden eagles, goshawks, and ospreys all nest in the area as well as numerous other raptors. The endangered shoetnosed sucker which inherits Gerber reservoir is given special consideration.

Numerous chances have been made in grazing practices over the past fifteen years and season of use and adjustments continue to be made. Grazing systems are in place which use multi-pasture rotations with short duration, high intensity grazing. This has allowed regrowth and rest of riparian areas and improved riparian condition. Use of these more intensive systems has not yet resulted in reductions in livestock forage allocations. Recently, two of the permittees in the "Gerber Riparian Demonstration Area" were Oregon state award winners and national finalists in the "Take Pride In America Awards."

Riparian restoration

The Gerber Block contains several miles of high quality riparian areas. This includes several intermittent streams which run significant flows in the spring of the year as well as many marshy areas and reservoir perimeters. They contain important nesting areas for waterfowl, ospreys, golden and bald eagles, and numerous other raptors in the area. In 1987 the area was designated a "Riparian Demonstration Area" where riparian projects would be showcased for interested individuals and groups. The objectives include restoration of stream flows, improvement of water quality, stabilization of stream channels, and restoration of streamside vegetation and wildlife habitat. Baseline and monitoring data are collected and changes and trends are documented.

I hope you're getting a picture of the area as an entire system of vegetative associations and the objective of general restoration across the landscape.

Prescribing fire

Enter! The return of fire to the system. Several large stand destroying fires have occurred in the area. The nearby Quartz Mountain Fire in 1977 on Fremont National Forest and the Bryant Mountain Fire on BLM and private land in 1979 prompted personnel to take a close look at what was happening with fuels on forests in the area. Prescribed fire was first introduced in 1980 and has continued every year since except 1982.

Table 1. Gerber Block Klamath County Oregon, acres underburned[1]

Year	Acres	Year	Acres
1980	3	1988	360
1981	20	1989	1675
1982	0	1990	1062
1983	35	1991	1142
1984	138	1992	1111
1985	306	1993	896
1986	100	1994	830
1987	450		

[1] *Goal is to burn 1500 acres per year.*

Experience was gained, participants are more knowledgeable, and between 1989 and 1994, 6,566 acres have been burned. Most burning has been done in April. This month has good air mixing, frequent light winds, cooler temperatures, and least danger of escaped fire. Some burning has been done in October but winds are not dependable and atmospheric mixing is not good.

There are four objectives to the burning program in the area:

- The reintroduction of fire to plant communities where fire historically had a profound influence.
- Restore sustainable function and structure to plant communities and improve forest health in fire adaptive systems.
- Reduce the threat of catastrophic wildfire resulting from heavy fuel loadings.
- Reduce overall fire management costs associated with large fires burning over many days.

Current observations

We are beginning to get some observations on past treatments, prescribed fires, and wildfire occurrence. Prior to underburning, fuel loads in ponderosa pine forests in the area averaged 61 tons per acre. Units underburned one time average 23 tons per acre. A wildfire in one of the once-burned units resulted in complete overstory loss and it is felt future treatments will be required to further reduce fuel loads. Two treatments will be necessary before we can get fuel loads to desired levels of around 12-14 tons per acre.

Personnel have conducted exhaustive literature searches in support of the program and have been allowed the latitude to innovate and try new things. They have worked closely with other state and federal agencies. Personnel from the Oregon Department of Forestry and personnel from Crater Lake Park have been especially helpful. They are now looking at the possibility of contracting some of the work including lining of relic trees, snags and down logs, patrolling, and control. Ignition will still be "carefully" done by agency personnel.

Impact of wild fire

We have had five wildfires on the resource area. Their impact is indicated on the following table.

Table 2. Klamath Falls Resource Area, intensity and severity

Name of Event	Intensity	Severity %
Bryant Mountain Fire - 1979	H	L(5),H(95)
Johns Spring Fire - 1992		
Area logged pre fire	L-M-H	L(30),M(60),H(10)
Area untreated	H	H
Area logged 1977 within proposed ACEC	H	M(20),H(80)
Area untreated in proposed ACEC- Presc.Nat.Fire	M-H	M(40),H(60)
Paddock Fire - 1992		
1989 Original Underburn	L	L
Wildfire in burned area (one burn only)	H	H
Kitts Mill Fire - 1987		
1986 Original Underburn	L	M
Wildfire in RX burn	M	M
Wildfire in non RX area (one burn only)	H	H
Fort Spring Fire - 1992		
1985 Original Underburn	L	L
Wildfire in RX burn	H	H
Wildfire in non RX area (USFS one burn only)	H	H
Gerber Underburns		
Spring 1989	L	L
Spring 1990	L	L(90),M(10)
Spring 1992	L	L(50),M(40),H(10)

In the case of the Johns Spring Fire in 1992, a stand destroying crown fire dropped to the surface upon reaching a treatment area which had been previously logged with an old growth retention prescription. There had been a partial removal of overstory with heavy commercial thinning from below. The harvest was heavy

in white fir stands. There is little argument that wildfire in treated stands, albeit when treated with practices which reduced fuels and fuel arrangement which supported stand destroying fire, have been more resistant to loss. Significant fuel reductions with at least two treatments will be necessary to assure stand survival.

The intensity of fire is not synonymous with severity. We have experienced fires that have appeared to be of low intensity, but due to extreme consumption of duff and concentrations of round fuels, have exhibited severe site and plant damage. Most of the prescribed fires to date were low intensity with low to moderate severity. Less then one tenth of the prescribed fire completed in 1992 (a very dry year) were classified as high severity. We have found that the lining of relic trees, snags, and down logs is necessary to their continued survival and/or existence.

Monitoring

To date there has been no research done on the Gerber burns. The monitoring of past and present burns is important and at least the following considerations are made. Before and after photo plots have been established with a desired return at 1,5 and 10 year intervals.

Items to monitor:

- Percentage of overstory cover and composition made up by individuals it senescence.
- Percentage of area meeting the Forest Service "Old Growth Definition." This defines the typical range (number) of snags, down logs, and trees within diameter classes.
- Changes in fuel models.
- Evaluation of protective measures on the health and/ or well being of relic and special specimens.
- Evaluation of impact of burning on special status species and habitats.
- Evaluate the savings of resources and funding that occurred as a result of the prescribed burn versus wildfire.
- Discuss new knowledge in fire effects that has been acquired because of the burning experience.
- Note changes in plant and animal communities and determining the cause.

Cost and other considerations

The cost of past prescribed burns in the Resource Area has averaged $40-50 per acre. Substantial increases are expected when dealing with isolated parcels and more complex fuel types. Maintenance burns are expected to cost about one half restoration burns. While some damage to resources is expected to occur, it has not appeared to be substantial. The cost of suppressing wildfires less than one acre in size is about $2,300 per fire. Larger fires in the area in 1992 ran from $1,800 to $3,000 per acre.

The Resource Area must comply with the Oregon Smoke Management Plan and local the Klamath volunteer smoke management process. There is considerable interest in the area in taking a new look at smoke management and wildfire smoke by accident versus prescribed fire smoke by choice. Several large wild fires in the area have given the issue some needed attention.

During preparation of the NEPA document, Klamath Falls Resource Area Environmental Assessment on Fire Management, consideration was given to such things as fire effect on vegetation, animals including big game and small animals, fish, amphibians, birds, soils, water quality, riparian areas, historic and cultural resources, recreation, and economics.

Conclusion

In times when criticism of land treatments seems to over-power science and reason, it is nice to be able to look back and see we have done some things right in the past and can adopt new knowledge and technology as we go along to promote sustainable natural systems. It is my opinion that two factors played, and are continuing to play an important roll in the success of the Gerber project. The first is a continuum of professional specialists who cared for the land, stayed in place long enough to see the changes and learn from them, and were not hesitant to try new and creative ways of doing things. The second is a history of line managers who continue to support these efforts and who made the sometimes difficult decisions required to get new things going. Personnel have been successful in building partnerships with most of the interested publics in the area.

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Lessons learned from fire use for restoring southwestern ponderosa pine ecosystems

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Abstract. Since European settlement, the southwestern ponderosa pine ecosystem has experienced large scale alterations brought about by heavy grazing and timbering and a policy of attempted fire exclusion. These alterations are most evident as large increases in tree numbers and in forest floor organic matter. These changes have resulted in forest health problems, such as increased insect and disease epidemics, reduced wildlife habitat, and a serious wildfire hazard. Prescribed burning used in ecosystem restoration can reduce heavy fuel accumulations, provide adequate microsites for natural pine regenerations, nonselectively thin dense stagnated thickets, and create an edaphic and stand environment conducive to better forest health and productivity. Research reported here indicates the improved forest conditions that result from burning. Conditions that more closely resemble those of presettlement will require other activities in association with fire.

With fire application, it is important to monitor conditions before, during, and after burning in order that positive fire effects can be replicated or adaptations to the prescriptions can be made.

Introduction

Prior to European settlement, the composition and structure of southwestern ponderosa pine (*Pinus ponderosa Laws. ex Doug.*) forests were quite different from today. The open, park-like presettlement stands, characterized by well-spaced older trees and sparse pockets of younger trees, had vigorous and abundant herbaceous vegetation (Biswell et al. 1973, Brown and Davis 1973, Cooper 1960). These forest conditions were maintained by naturally-ignited fires burning on a frequent, regular basis in light surface fuels of grass and pine needles. Light surface fires burned at intervals averaging less than 10 years and as often as every 2 years (Dieterich 1980, Weaver 1951). Warm, dry weather common to the Southwest in early summer, the continuity of grass and pine needles, and the high incidence of lightning caused this short fire interval. Light surface fuels built up sufficiently with the rapid resprouting of grasses and the abundant annual pine needle cast. Large, woody fuels in the form of branches or tree boles, which fall infrequently, rarely accumulated over a large area. When they were present, subsequent fires generally consumed them, reducing grass competition and created mineral soil seedbeds which favored ponderosa pine seedling establishment (Cooper 1960). These effects created an uneven-age stand structure composed of small, relatively even-aged groups.

The decline of the natural fire regime in southwestern ponderosa pine ecosystems started with extensive livestock grazing in the late 19th century when fine, surface grass fuels were reduced (Faulk 1970). Subsequently, ponderosa pine regeneration increased because of reduced understory competition, less fire mortality, and more mineral seedbeds (Cooper 1960). In the early 1900's, forest practices, primarily fire suppression, further reduced the ecological role of fire. These practices lead indirectly to stagnation of naturally regenerated stands and unprecedented fuel accumulation (Biswell et al. 1973).

Stand stagnation has been reported on tens of thousands of acres throughout the Southwest (Cooper 1960, Schubert 1974), and still persists where natural or artificial thinning has not taken place. Sites with dense thickets are not only unproductive but also represent a severe wildfire hazard.

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For several decades, trees of all sizes in untreated stands have been showing signs of stress with generally poor vigor and reduced growth rates (Cooper 1960, Sutherland 1983, Weaver 1951). This condition, which often results in increased insect and disease attack, is likely due to reduced availability of soil moisture caused by intense competition and by moisture retention in the thick forest floor (Clary and Ffolliott 1969). Thick forest floors also indicate that site nutrients, especially nitrogen, may be limiting because they are bound in unavailable forms (Covington and Sackett 1984, Covington and Sackett 1992).

During the last 75 to 100 years with a greatly altered natural fire cycle, unprecedented and unnaturally large amounts of surface and ground fuels have accumulated (Kallander 1969). Sackett (1979) reported average loadings of naturally fallen fuels at 22 tons per acre for 62 southwestern ponderosa pine stands. Harrington (1982) verified the heavy fuel loadings with an average of 34 tons per acre in southeastern Arizona.

Forest floor fuels approaching 9 tons per acre have been measured in sapling thickets and more than 50 tons per acre on old-growth sites. Annual fuel accumulation on those sites can range from 0.6 tons per acre to more than 3.5 tons per acre (Sackett and Haase in preparation). The decomposition rate (k) (Jenny et al. 1949) in these forests is extremely slow, resulting in the large buildup of forest floor fuel. K values range from 0.076 to 0.059 and 0.050 per year for sapling, pole, and old-growth substands respectively (Sackett and Haase, in preparation). These rates are less than some of the slowest decay rates reported in the literature which are 0.18 and 0.08/year in young stands and old-growth stands of ponderosa pine, respectively, in northern California (Hart et al. 1992).

Large, woody fuels, formerly uncommon in the Southwest, now average about 8 tons per acre but are frequently found at twice that loading (Sackett 1979). Much of the heavy fuels have accumulated in sapling thickets, further increasing the potential for crown fires.

A combination of heavy forest floor fuels and dense sapling thickets, coupled with the normally dry climate and frequent lightning- and human-caused ignitions, has resulted in a drastic increase of severe wildfires in recent decades (Biswell et al. 1973, Harrington 1982). Data summaries from USDA Forest Service Smokey Bear Reports in Figure 1 show a great increase in the number of acres burned by wildfire in Arizona and New Mexico since 1970. Of all the years since 1915, with over 100,000 acres burned, almost 70 percent occurred between 1970 and 1990, indicating a worsening problem.

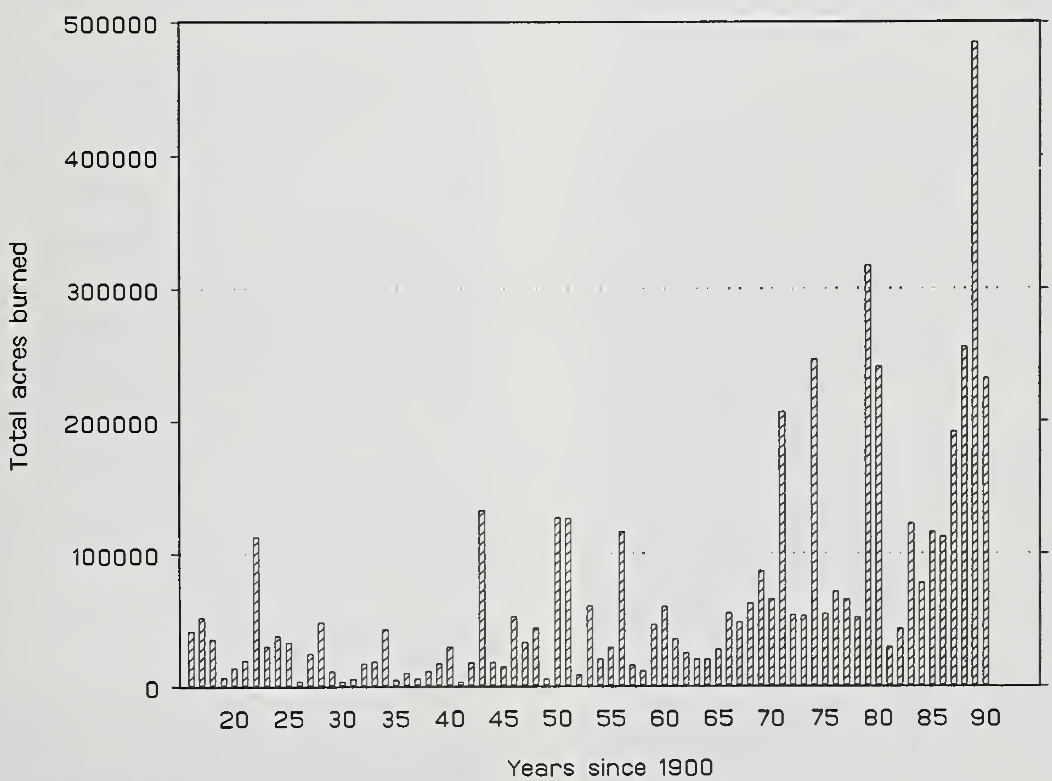

Figure 1. The total number of acres burned as wildfires in Arizona and New Mexico from 1916 to 1990. Data obtained in USDA Forest Service Smokey Bear Reports.

A final characteristic of the present southwestern ponderosa pine stands is the sparseness of understory vegetation, including pine regeneration. The thick organic layers and dense pine canopies have suppressed shrubby and herbaceous vegetation (Arnold 1950, Biswell 1972, Clary et al. 1968). In openings left by overstory mortality where pine regeneration is desired, conditions for establishment are poor, again because of deep organic layers which quickly dry out (Haase 1981, Sackett 1984). This condition has reduced the wildlife, range, and timber production value of these forests and has generally resulted in minimal biodiversity.

Reintroducing fire as an ecological process

Because natural fire was the major presettlement factor in shaping and maintaining southwestern ponderosa pine ecosystems, it is logical to consider applied fire in a management scheme to relieve the serious problems that plague these forests due to years of fire exclusion. Fire has been used in the southeastern United States for many years to maintain pine on sites that would successively shift to hardwoods without fire. In the Interior West it is also recognized as the key factor in keeping healthy, seral ponderosa pine stands from becoming stressed and prone to severe wildfire because of increases in tree density and changes in species composition (Arno 1988).

The hazardous conditions described make the widespread application of prescribed fire difficult and costly. Heavy fuels and dense stands can create control problems, and mortality of the stressed overstory from anticipated above- and below-ground heating is a certainty. An acceptance of these risks and economic losses, however, seems necessary in the short-term if ecological sound management is sought.

For prescribed fire to be effective in reducing the fire hazard and improving stand health, consideration must be given to a sequential burning program which would somewhat replicate presettlement fire activity (Sackett 1975, Sackett 1980b). One burn typically will not be sufficient to reduce fuels and stand density initially and certainly not over the long term. Frequently, the fire hazard remains high after one application because of the addition of fire-killed fuels (Harrington 1982). To be effective, maintenance burning is necessary to keep recurring fuels to a minimum (Davis et al. 1968, Gaines et al. 1958, Harrington 1981, Sackett 1975, 1980b). Generally, repeat burns in light, needle fuels are easily manageable.

Historically, natural fire in presettlement times probably burned during the period just after the spring dry season, just as the first storms developed announcing the start of the monsoon season in the Southwest. These first storms are typically dry, and the accompanying lightning could start numerous fires. With the exceptionally high fuel loading and dense stands of today, spring would not be the preferred season for the initial fire entry because the most severe part of the wildfire season is imminent. Fuel reduction and overstory thinning should be accomplished in stages over time. Fall, then, becomes the season of choice when weather and fuel moisture conditions are more moderate, and high winds not as likely. Once stands have been conditioned with one or two effective fuel reduction burns, spring burning becomes a more realistic option to lengthen the burning season. Summer prescribed burning can also be successful as an alternative to fall when conditions are often poor for burning (Harrington 1981, 1987).

The real premise of prescribed fire use in fire adapted ecosystems is to provide for interval burning on a rotation that promotes healthy, wildfire-resistant, productive forests.

Two examples

In 1976 and 1977, companion studies were established near Flagstaff, Arizona, to investigate the effects of reestablishing fire in ponderosa pine. Study areas were established on the Fort Valley Experimental Forest in 1976 on a basalt soil site now referred to as Chimney Spring. One year later, a research site was established on the Long Valley Experimental Forest on a limestone/sandstone soil now known as Limestone Flats (Sackett 1980b).

The initial objective of these sister studies was to determine a burning interval that would adequately manipulate fuels and tree density in an overstocked post settlement ponderosa pine stand so that it would survive a wildfire that would otherwise be stand-replacing. The hypothesis of this research was that reestablishing prescribed fire as a surrogate to natural fire would be beneficial to southwestern ponderosa pine ecosystems. The primary focus of the study was to deal with the most apparent problem in the pine ecosystem, that of unnaturally heavy forest floor fuels.

Fuels and fire behavior

Initially, loadings of forest floor fuels, which includes all organic matter less than 1 inch in diameter, in both Chimney Spring and Limestone Flats were similar, 15.2 and 15.7 tons per acre, respectively. Limestone Flats had more than 16 tons per acre of woody fuels greater

than 1-inch diameter, whereas Chimney Spring had about 7 tons per acre (Sackett 1980b). The importance of fuel moisture on fuel consumption and, therefore, fire effects was demonstrated when all the interval burning treatment plots (1-,2-,4-,6-,8-, and 10-year) were initially burned in 1976 at Chimney Spring and in 1977 at Limestone Flats. A dry summer and fall in 1976 resulted in low fuel moistures so that the initial burn at Chimney Spring was done at night when the humidity was higher and temperatures were lower. As a result, 63 percent of the forest floor fuel was consumed, as was 69 percent of the woody fuels. In contrast, the Limestone Flats area was burned in the fall of 1977 after a wet summer and fall. As a result, only 42 percent of the forest floor material and 44 percent of the woody fuels were consumed.

Annual burning (1-year interval) is a rotation established to determine the feasibility and effects of such frequent burning. We have found that annual burning is not possible, not because of insufficient fuels to carry a fire, but because weather and fuel moisture in certain years are too damp. Increased windspeeds can sometimes compensate for marginally damp fuel conditions, allowing fire to carry in these light fuels.

Burns attempted on a 2-year rotation are generally more successful because of the slightly greater fuel loads. Again, marginal weather in fall makes biennial burning dubious. Biennial burns may be effective in wildland/urban interface situations, where precise control is important.

The most effective prescribed burning rotation observed at Chimney Spring is the 4-year interval. Although this rotation has burned well each interval and has not damaged the healthy overstory, it is not certain whether optimal weather has occurred synchronously with those years or if 4 years is the optimum burning cycle. This rotation appears to be effective because of the consistent ease of carrying out the treatment in keeping fuels to a minimum. To test whether each rotation meets the objective of reduced wildfire hazard, heading fires are ignited for each burn to determine if the stand is protected from a wildfire situation. To date, 4-year rotation burns have done well to meet that objective.

Six-year burning rotations begin to accumulate fuel loads that stretch the fire intensities to an upper limit that may cause undesirable damage to the residual overstory. The two times that the 6-year intervals have been burned, after the initial burn have yielded contrasting results. But, fuel loads are such that under severe fall fire weather conditions, fires could be a control problem and lead to undesirable fire effects.

An example of this condition occurred in the fall of 1992 which was warm and dry, and frequently windy. With 42 rainless days, the heavy, woody fuels had thoroughly dried out from the summer monsoons. Rotations of 1-,2-,4-, and 8-years were burned at the same time. All except the 8-year rotations burned well and did not result in excessive crown scorch. However, with 8 years of fuel accumulation (5 tons per acre), low fuel moisture (4 percent to 6 percent), low humidity (21 percent), and only moderate winds, a 1-chain deep strip heading fire heavily scorched most of the pole and smaller size trees in a one-half-acre area. Continuing with heading fires would have resulted in extensive stand mortality. By allowing the fire to continue as a backing fire well into the night, the 8 years of fuel accumulation was safely consumed. The severity of this 8-year interval burn points out clearly the need for continuous, short-interval burning in an ecosystem sustained by fire.

The only test of a 10-year burn interval occurred in 1986. Fall conditions were too damp for effective fire spread. Forest floor fuel had accumulated in 10 years to more than 7 tons per acre, so experience from the 8-year burns would suggest severe overstory damage would have occurred if conditions had been warm and dry; and a freely spreading surface fire was allowed.

Monitoring fuel loading and consumption, burning conditions and fire behavior are all essential parts of learning from the adaptive management process of using prescribed fire. Before establishing a prescribed fire program on a particular site, forest floor fuel loadings can be determined easily in undisturbed, natural fuels using depth-loading relationships (Sackett and Haase, data on file). Using prediction equations, depth measurements can be converted to fuel loadings. Measurements after the burn can be subtracted from preburn measurements to determine fuel consumption. A simple sampling grid system can be used to measure the depth. Sampling should be stratified by stand type (old-growth, sapling thicket, large pole, etc.) because of differences in forest floor bulk density. Knowledge of the amounts and fate of woody fuels is also important and should be monitored using the planar intercept sampling scheme (Brown 1974).

During the fires, weather and fuel moisture conditions should be monitored allowing comparisons, correlation, or prescription modifications. Fuel moisture is typically measured using Nalgene sampling bottles, a balance, and a standard drying oven. Other rapid response moisture analyzers are also available (Norum and Fischer 1980, Sackett 1980a). Moisture content sampling should be stratified to show expected differences from different crown closure conditions, fuel depths, layers, and/or geographic location. Measurements or estimates of rates-of-spread and flame lengths can be important in assessing fire impacts and methods as described in Rothermel and Deeming (1980).

Natural pine regeneration

Regeneration of ponderosa pine has obviously been sufficient to perpetuate the ecosystem for thousands of years. Except in isolated situations, attempts to regenerate southwestern ponderosa pine stands naturally or by direct seeding have failed (Heidmann et al. 1977). Schubert (1974) identified several coincidental conditions necessary for successful regeneration of ponderosa pine. In the past, fire functioned to prepare competition-free, mineral microsites that gave the highest probability for pine seedling establishment. Prescribed fire can provide mineral soil seedbeds for superior germination and early growth.

Especially at Chimney Spring and to a lesser extent at Limestone Flats, natural regeneration and seedling survival have been satisfactory. As a result of the initial burns at Chimney Spring, mineral soil was exposed on 19 percent of the area, mostly around large, old-growth trees and where rotten logs were consumed (Sackett 1980b). Seedlings began to appear soon after summer rains started in the year succeeding these burns and were concentrated in areas where forest floor consumption was sufficient to expose mineral soil (Sackett 1984). First inventories made in August, 1977 indicated that an equivalent of 2,600 seedlings per acre were present on the 18 burned plots. Seedlings excavated on burned sites had long tap roots, allowing them to avoid desiccation from fall drought and to resist frost heaving, giving them a survival advantage. Roots of seedlings in unburned plots generally failed to significantly penetrate the heavy forest floor and into mineral soil, resulting in high fall and winter mortality.

Many of the 1977 seedlings have been killed by subsequent fires, but of the survivors, many are now 4- to 8-foot tall saplings. The trees that have survived are found on sites where large, old-growth trees were killed by the initial burns. On these sites, fine needle fuels have not been available for fire spread. Obviously, these are the very sites where pine regeneration is desired. Where pine litter fuels are produced, stocking is generally adequate, so the killing of seedlings with subsequent fire is inconsequential and, in fact, desired to prevent overstocking.

Since 1976, there have been two other good seed years where seedlings have flourished at Chimney Spring. On one burned plot, the equivalent of 650,000 seedlings per acre were counted (Sackett and Haase, data on file). Seedbeds remain viable for up to 7 years after a fire (Sackett and Haase, data on file). Needles cast during this interval form only a loose, highly porous mat which allows seeds to fall through and settle on mineral soil (Haase 1981). Without fire as a natural disturbance to the forest floor, pine regeneration will only be successful with highly impactive and ecologically questionable mechanical scarification.

Again, monitoring is the key to verifying the success of restoration efforts. If mutual regeneration is desired, monitoring cone crops and burning to coincide with a good crop would be beneficial. Then surveys of seedling survival and growth on various microsites would assist in future forest planning.

Thinning of stands

At Chimney Spring many of the smaller overstory trees were severely crown scorched and, as desired and expected, died. The crowns of the large, young-growth and old-growth trees, however, were not greatly affected. A noteworthy observation was that the forest floor from the bole to the dripline around each of the 405 large trees was completely consumed (Sackett and Haase 1992). About 1 1/2 years after the burns, a number of large pine began to fade and die. None of the same size trees on control plots exhibited these symptoms. Since the crowns were not scorched during the 1976 fires, other living parts of the trees, roots and cambium, sustained damage. To date, 39 percent of the large, overmature trees are dead on plots burned, whereas 16 percent are dead on unburned, control plots (Sackett and Haase, data on file). Increased mortality on burned plots can be attributed to the initial burns. This example establishes the desirability of long-term fire effects monitoring, not just immediate effects. Long-term monitoring is needed to verify changes attributed to the adaptive-management treatment.

A major role of natural fire in the presettlement era was the thinning of young trees, giving the landscape the open, park-like character. The dense structure of southwestern ponderosa pine forests today, which is the primary cause for poor forest health and severe wildfire hazard, forces managers to consider alternative methods of thinning. Much of our forest lands are thinned mechanically, but prescribed burning can also be effectively used as a thinning agent. Because of the frequency of presettlement fires, most trees were killed in the seedling stage by the common low intensity fires. Where heavy fuels and fallen trees burn out, seedlings are able to become established because of the elimination of fuels and competition. Using prescribed fire within stands as they exist today is different because the stands needing thinning are dense "dog hair thickets" of 80-year old pine saplings. Although the saplings in these thickets are of small diameter due to close spacing and competition, the bark is disproportionately thick. Prescribed fires in those thickets are usually not as intense as in open stands. Shade, higher fuel moistures, and minimal amounts of humus in the forest floor prevent

lethal temperatures from being produced around the tree bases. We have found that crown scorch and/or consumption in excess of 75 percent is necessary to thin dog hair thickets.

Initial burns at Chimney Spring reduced the number of stagnated reproduction and sapling stems from an average of 1553 to 912 stems per acre (Harrington and Sackett 1990). Small poles, many of which are also stagnated in thickets, were reduced from 192 to 156 stems per acre. As mentioned previously, burning at Limestone Flats was less effective due to wet conditions; an average of only 180 stems per acre were killed by the fire in reproduction/sapling size classes.

In an undisturbed, well-developed forest floor, newly cast needles become rapidly colonized and bound by mycelium and therefore less burnable. On a burned site, pine litter that falls after a fire does not become readily bound by mycellia and a much deeper layer of pure litter accumulates. Under good burning conditions, repeat fires consume most of the needles and small twigs in a flaming front, so that fire behavior, rates-of-spread, fire intensity, and flame lengths are much higher in response to the greatly increased amount of available fuel. This increased fire behavior potential can be used to purposely increase crown scorch in stagnated thickets.

At both prescribed fire research areas, thinning of dense stands has been an objective to relieve the stagnated condition. The ability to manipulate the fire through ignition techniques and the fire environment to achieve slow-dissipating, high temperature air in the crowns is necessary to use fire as a thinning tool (Harrington and Sackett 1990, Sackett 1968). Adjusting the direction of fire spread relative to windspeed is the most common technique. Heading or uphill fires move at a speed commensurate with windspeed creating more intense fire behavior. On the other hand, backing fires, moving against the wind (or downhill), progress with short flames and low intensities, and seldom thin stands. Season of burning can also have an affect on thinning. Burning at times of the year when tree susceptibility to fire damage is high, adds other dimension to thinning with fire. Although fall burning is recommended for initial burns, repeat burns might well take advantage of spring and summer conditions for thinning when rapid growth leaves trees susceptible to scorching (Harrington 1987).

Skillful manipulation of prescribed fire techniques and conditions is required to thin dense ponderosa pine thickets. It is, however, another way prescribed burning can be used to relieve unnatural conditions in a fire-dependent ecosystem.

Establishing permanent sample points with stem counts in sapling thickets is a simple method to monitor changes in stem numbers to evaluate the success of restoration attempts. Where concerted efforts are underway to reduce the number of sapling stems, permanent photo points provide a valuable record of how stand structure and composition change with each successive burn.

Understory vegetation responses

In southwestern ponderosa pine forests, understory vegetation has declined steadily from the presettlement era. The decline has long been attributed to the exclusion of fire, the subsequent increase in heavy forest floor accumulations, and increased overstory densities (Cooper 1960, Biswell 1972). Burning at Chimney Spring and Limestone Flats has resulted in substantial changes in the understory. Most evident is the abundance of disturbance invader species like Mullein (*Verbascum thapsus* L.), toadflax (*Linaria dalmatica* L. Mill), and thistle (*Circium pulchellum* [Greene] Woot and Standl.). Mullein and toadflax are dominant on severely burned sites around fire-killed, old-growth trees. Although some animals use these plants (Patton and Ertl 1982), none are considered favored by wildlife or cattle.

Grass species respond to prescribed fires and wildfires differently as noted throughout the literature. Generally, production is increased, but this depends on fire severity, season of burn, and overstory characteristics. Individual species will also respond differently. Arizona fescue (*Festuca arizonica* Vasey.) and squirrel tail (*Sitanion hystrix* [Nutt.] J.G. Smith) usually show an increase in production 1 year after a fire (Harris and Covington 1983, Sackett and Haase, unpublished data, Vose 1984) whereas mountain muhly *(Muhlenbergia montana* [Nutt.] Hitche.) requires a longer recovery period.

In 1992, a vegetation survey was made at Chimney Spring study area on the control, 1-, 2-, 4-, and 8-year rotation before burning. Preliminary review of the data substantiates previous research. Production of mountain muhly and buckbrush (*Ceanothus fendleri* Gray) was reduced immediately following the prescribed burn. On the 4-year interval plots after four consecutive burnings, mountain muhly had almost recovered to the level of the control plots (46 observations on burned plots, 53 observations on control plots), and the 8-year rotation plots after two burns had a much greater number of observations (92-burned, 53-control). The 2-year-interval plots showed a small increase in number of observations from the 1-year-interval plots (38 and 32 respectively). Buckbrush appears to require a longer recovery time also. The 1-,2-, and 4-year rotations had substantially fewer observations (6, 2, and 6 respectively) than the 8-year rotation and the control plots (17 and 19 respectively), but this may be due to the reoccurring fire treatments.

These data reflect density differences between burning treatments. Evaluation by cover class should

show that overall biomass production is greater in the burned plots because plants were visibly larger than those in the control plots. Much of the current vegetation response research takes into consideration the effect of the small, even-aged groups of ponderosa pine (Harris and Covington 1983, Oswald and Covington 1984, Vose 1984). The greatest vegetation response occurs in open, mature timber stands or directly beneath the mature timber canopies. Generally, little change in vegetation is seen in pole stands or in the dense sapling stands. This further indicates the need for stand density reductions.

Photo records over time are a good way to visually judge success of fire use in restoration and can complement quantitative methods which are necessary in evaluating these effects. Many sampling schemes are available to monitor vegetation changes. It is important that the sampling be done at an adequate scale, done each year at the same physiological time of the major plant species, and done at permanently established points.

Conclusions

Very few forest ecosystems compare with southwestern ponderosa pine in the frequency of presettlement fire. Written and photographic chronicles indicate the importance of this fire regime for maintenance of forest health and stability (Harrington and Sackett 1990). In the past 75 to 100 years of attempted fire exclusion, these forest qualities have greatly degenerated. Prescribed fire, then, should be considered as a primary agent for restoring ecosystem health and biodiversity similar to the presettlement condition. Additionally, a more sustainable ecosystem should result with the reduction of severe wildfire potential.

After 18 years of burning on two sites with greatly altered vegetation and organic layer compositions, it is evident that presettlement conditions or a desirable likeness would be difficult to achieve with prescribed fire alone. The monitoring scheme has allowed this evaluation to be made and points towards activities that may include a fire program supplemented by low-impact mechanical thinning.

The intent of restoration ecology is to return an altered ecosystem to one that exhibits functioning, natural processes and the products of those processes. In southwestern ponderosa pine forests, fire should be viewed as a distinct ecosystem process and not simply as an alternative management tool. Its direct and indirect impacts on key parts of the forest (vegetation, forest floor, and soils) are unique and can determine the quality of the ecosystem. The use of ecologically sound, adaptive management complete with detailed, long-term monitoring should increase the probability of achieving desired future conditions.

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Thinning as a tool in restoring and maintaining diverse structure in stands of southwestern ponderosa pine

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Abstract. Thinning is a silvicultural tool which can be used to modify current stand structure and promote a more balanced structure in areas with either excess small trees or currently lacking a large tree component. Periodic thinnings can also maintain tree and stand vigor while meeting structure goals, providing managers select silvicultural options with adequate consideration of the interactions with insects and diseases.

Introduction

Conditions in large areas of ponderosa pine (*Pinus ponderosa* var. *scopulorum* Engelm.) forests in the Southwest have been greatly altered by a long history of heavy grazing, timber harvesting, and fire suppression (Cooper 1960; Covington and Moore 1994; Weaver 1951). These agents have markedly changed forest and stand structure from conditions before Euro-American settlement. With the conversion of many stands to second growth management, large old tree components have been removed or significantly diminished. In other areas, populations of small trees which became established earlier in this century now dominate the stocking of forest stands. The effect has been to largely reduce stand structure diversity and increase stand density (Covington 1994; Johnson 1995). This paper presents methods based largely on the principles of uneven-aged management which can be used to improve structural diversity of forest stands (MacCleery 1995). Growth responses from on-going levels of growing stock studies in even-aged stands are used to infer growth responses possible when stands are managed using group selection cutting methods. While methods presented here can be used to create stand structures similar to known or inferred conditions of a century or more ago, the primary goal is to contribute to development of a flexible "naturalistic silviculture" to construct stand conditions which meet management objectives determined by society (Smith 1994).

Stand structure

Stand structure in the context of uneven-aged management refers to diameter distribution characterized as the numbers of trees by d.b.h. classes. Size distribution is usually related to age distribution, but the relationship may be misleading for a given stand (Foiles 1978). Examination of age structure at the Gustav A. Pearson Natural Area (GPNA) at Fort Valley Experimental Forest demonstrated that groups of presettlement trees which were relatively even-sized were clearly not even-aged (White 1985). In contrast, Cooper (1961) observed a more even-aged group structure in the White Mountains of Arizona. Varying patterns of age structure are the result of great variation in successful establishment of ponderosa pine in response to erratic seed production, climatic variability, and lack of uniformly good regeneration sites over large areas (Pearson 1950; Schubert 1974). Ponderosa pine in the Southwest has been demonstrated to grow well in both managed even-aged stands and in uneven-aged stands managed with the group selection cutting method, where groups normally range in size from 0.25 to 2 acres (Schubert 1974). This silvicultural flexibility is a management asset as it allows for uneven-aged management at the stand level where conditions are appropriate. Even-aged management within multiple stands can also be practiced to produce uneven-aged forests.

Stand structure in unmanaged and previously uncut stands is often relatively diverse, and while structure can be described as uneven-aged, examples of balanced

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uneven-aged diameter distributions are rare (Pearson 1950; Schubert 1974). It is unlikely that an actual stand will have a smooth balanced diameter distribution; however, the idealized uneven-aged structure provides a model to guide future management (Foiles 1978). The most popular theory of uneven-aged stand structure is the familiar inverse J-shaped curve of numbers of trees plotted by successively larger d.b.h. classes. The shape of the curve is determined by a "q-ratio" which is calculated as the ratio of numbers of trees in successively larger d.b.h. classes. Methods for determining the q-ratio and the inverse J-shaped curve are presented in detail by Alexander and Edminster (1977). A lower q-ratio produces a flatter distribution with greater numbers of larger trees, and a higher q-ratio produces a steeply declining curve with greater numbers of small trees.

Specifying uneven-aged management treatments involves determining a desired residual stand density and the shape of the inverse J-shaped curve. These diameter distribution goals can be developed from reference conditions from earlier studies. Cooper (1960) describes two unmanaged stands representative of conditions before white settlement and wildfire control with basal area densities of 66 and 86 square feet per acre. Basal area in the GPNA averaged 62 square feet per acre at the time of the initial inventory in 1920 (Avery et al. 1976). All of these values are for trees 4 inches d.b.h. and larger. These densities are within the range of standards often applied in local even-aged management. At the time of the initial inventory in GPNA, seedlings resulting from the regeneration years of 1914 and 1919 (Pearson 1950) were not recorded. These seedlings have since developed into the dense understory which limits the introduction of prescribed fire without initial thinning and reduction of fuels (Covington 1994). Previous conditions in the three stands in these studies can also be used to determine the shape of a desired diameter distribution. Q-values in the stands for 2-inch d.b.h. classes were determined as 1.22, 1.34, and 1.24. These values equate to q-ratios for 1-inch d.b.h. classes of 1.10, 1.16, and 1.11, respectively. For 4-inch classes, the q-ratios are 1.48, 1.80, and 1.54.

Applying stand structure goals in the GPNA

Withdrawal from natural area status of approximately 8 acres of the GPNA next to the Fort Valley Experimental Forest headquarters compound as a fuelbreak provided an opportunity in 1992 and 1993 to demonstrate implementation of thinning treatments to approximate conditions which existed at the time of the first inventory in 1920. The target residual stand structure for the fuelbreak area was determined by plotting the diameter distribution in 1920 and developing a inverse J-shaped reference curve which roughly described the distribution of trees in 4-inch wide d.b.h. classes (upper graph in fig. 1). The lower graph in figure 1 shows the distribution in terms of basal area stocking with the transformed reference curve. This original distribution is not balanced with surpluses of trees in the 6, 18 through 30-inch d.b.h. classes, and deficits in others. Nevertheless, the J-shaped diameter distribution curve provides an adequate guide for treatment. This curve was then applied to the diameter distribution for the area in 1992 (upper graph in fig. 2). The distribution in 1992 shows the dense condition of trees in the 14-inch and smaller d.b.h. classes. The distribution in terms of basal area (lower graph in fig. 2) further demonstrates the overstocking of smaller diameter trees.

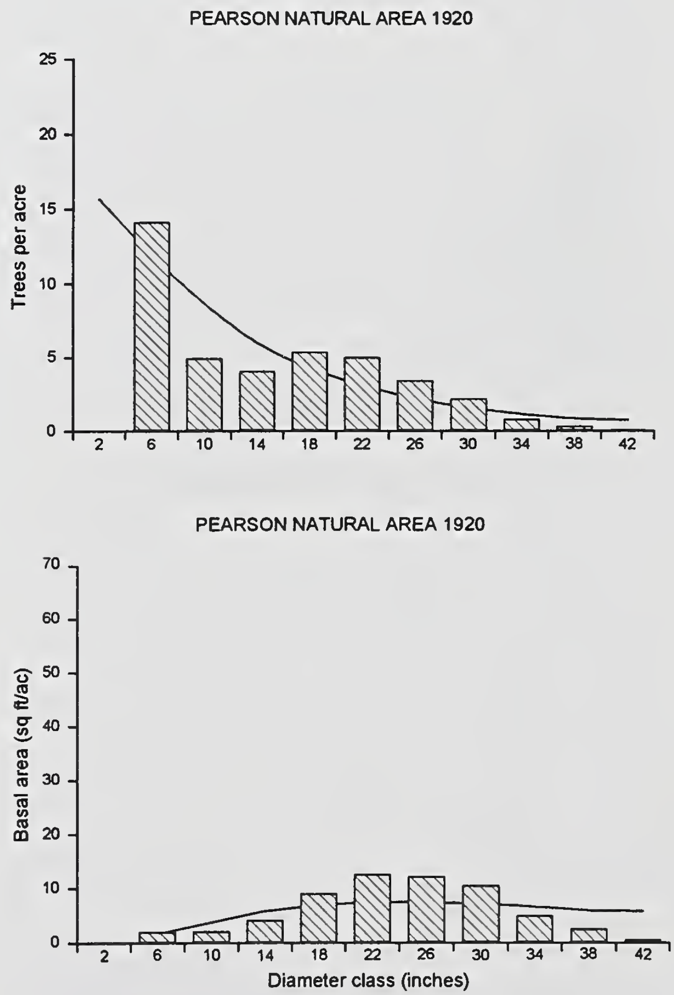

Figure 1. Diameter distribution in the Pearson Natural Area, 1920, in terms of trees per acre (upper graph) and basal area (lower graph) with the reference inverse J-shaped curve.

A prescription was developed which involved removal of surplus trees (numbers above the reference J-shaped curve) smaller than 12 inches d.b.h. Most trees in the 14-inch class were retained to balance deficit stocking in the 18- and 22-inch classes. Residual small trees were retained in small groups to represent the patchiness of the mature overstory. Note that the reference J-shaped guide was not applied in a rigid manner as surplus trees in the 30- and 34-inch classes were also retained. Resulting basal area density in the area is estimated to be slightly over 70 square feet per acre after treatment. Treatments deviated from normal operational practices in that all cut trees were removed from the site. Duff and litter accumulations around large trees were also removed before implementing prescribed burning treatments in the area. These initial treatments are just the beginning of a series of periodic treatments to maintain an uneven-aged structure at densities near target levels which promote vigorous tree growth. The effect of these treatments on the vigor of old-growth trees is being monitored. Stand conditions in the younger and mid-aged components are dynamic, as evidenced by growth rates discussed in the following section.

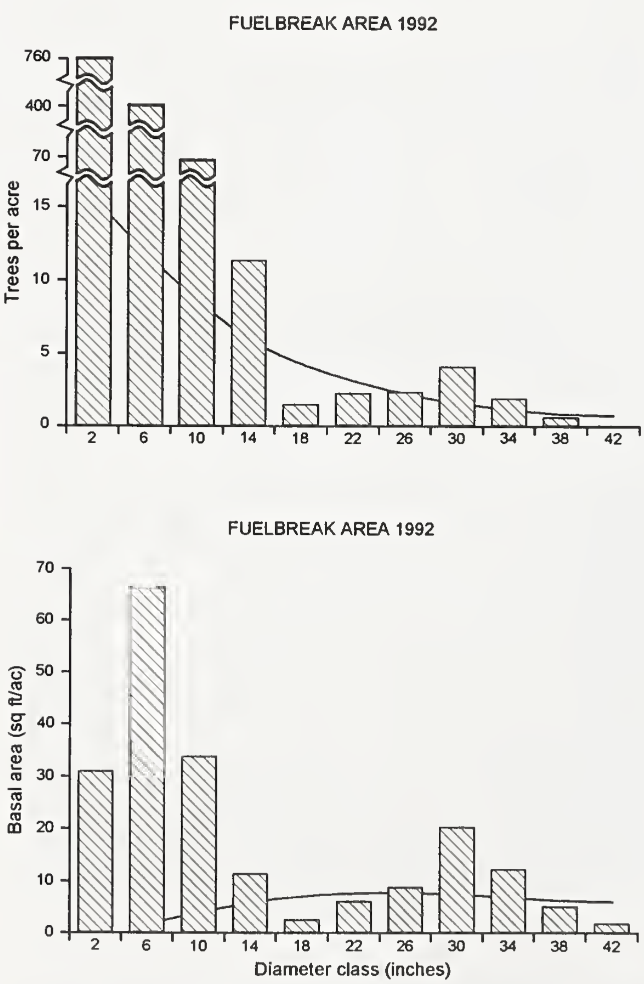

Figure 2. **Diameter distribution in the fuelbreak area, 1992, in terms of trees per acre (upper graph) and basal area (lower graph) with the 1920 reference inverse J-shaped curve.**

Enhancing structural diversity in second-growth stands

Stand conditions in the fuelbreak area before treatment actually present a situation where developing a management prescription is relatively straightforward given the original uneven-aged character of the stand. More challenging situations are areas where the mature tree component has been harvested and a conversion to second-growth management is well underway. In many cases, past implementation of prescriptions for these areas have resulted in relatively homogeneous tree size distributions and spatial arrangements aimed at maintaining vigorous tree growth and stand volume production. Large areas of the southwestern ponderosa pine type are in this condition. Properly applied thinning prescriptions can greatly improve both spatial and structural diversity in these stands and provide a means for developing an uneven-sized, if not uneven-aged, distribution.

The most effective means of creating a diversity of tree sizes in even-aged stands is by providing a variety of density levels for residual trees in a grouped spatial arrangement patterned after the arrangement of stumps from harvested large trees. As in the application of the group selection cutting method in uneven-aged silviculture discussed above, group size would normally be expected to be in the range of 0.25 to 2 acres. In the relative short run, a goal is to develop a more diverse diameter distribution which approximates an uneven-aged structure. Only time can allow the development of old trees, but varied density management can result in diversity of tree sizes in a relatively short period of time. In the longer run, stand conditions must also be favorable for establishment of periodic regeneration in group-sized areas. This will mean creating openings and maintaining them with prescribed burning until regeneration is established. Prescribed burning can also aid in controlling density of established regeneration (Harrington and Sackett 1990; Wright 1990).

Results from the Taylor Woods level of growing stock study (Ronco et al. 1985) at Fort Valley Experimental Forest provide a framework to assess the effects of density management on tree growth and stand development in second-growth stands and in the small and middle diameter range in uneven-aged stands with a

grouped pattern. While not directly applicable to uneven-aged stands, the results provide insights into growth responses at smaller even-aged group scales. Average plot size at Taylor Woods is close to the mid-range of group size of about an acre. The Taylor Woods study began in 1962 in a stand of mostly 43-year-old small diameter saplings and poles. Site index for the area averages 73 feet at base age 100 years (Minor 1964) and is somewhat above the regional average for ponderosa pine. Plots within the study area have been thinned to one of six residual treatment levels at 10-year intervals since 1962. Levels of stocking being examined are 30, 60, 80, 100, 120, and 150 square feet of basal area per acre. In 1992, the residual level on the most open plots was raised to level 40 to maintain site occupancy on plots with the largest diameter trees. Each treatment level is replicated on three plots. Details on the installation are provided by Ronco et al. (1985). The discussion which follows is based primarily on results from the first 20 years of the study; however, preliminary analysis of the latest period from 1982 to 1992 show trends consistent with the earlier time periods.

The dynamic growth responses at the tree and stand level are the most dramatic results of the study. Stand density management across a wide range of residual density retained after thinning is demonstrated to be a powerful tool in affecting growth rates and resulting sizes of trees. Periodic annual average diameter growth ranged from 0.34 to 0.10 inches with increasing residual basal area (fig. 3). Average diameter growth is strongly negatively correlated with residual basal area. The growth rate at the lower stand density is approximately 3 times greater than at high densities. In just 30 years since the initial treatment, trees on the open plots average nearly twice the diameter of trees on the densest plots. Average annual height growth has also shown a decreasing trend with increasing residual basal area, but the effect of increasing density is less generally less than 0.3 feet and not nearly as significant as average diameter response.

Periodic annual basal area increment ranged from 2.2 to 4.5 square feet per acre with increasing residual basal area during the 1963 to 1972 period and from 1.9 to 4.0 square feet per acre during the 1973 to 1982 period (fig. 4). Basal area increment was consistently greater during the initial adjustment period. Results from the latest period show a consistent response with the second period except for a slight decrease in growth rates at open stand densities. Growth rates of 20 to over 30 square feet per decade over the range of low to high densities will require frequent entries to maintain group and stand densities in a low to moderate range. The development of basal area stocking is one of the most obvious measures portraying the dynamic growth responses of young and mid-age stands.

Periodic annual merchantable volume increment ranged from 34 to 77 cubic feet per acre with increasing residual basal area during the first period and from 34 to 75 cubic feet per acre during the second period (fig. 5). Growth rates for the most recent period lie between the two earlier periods. Merchantable volume production should allow periodic commercial removals over as short

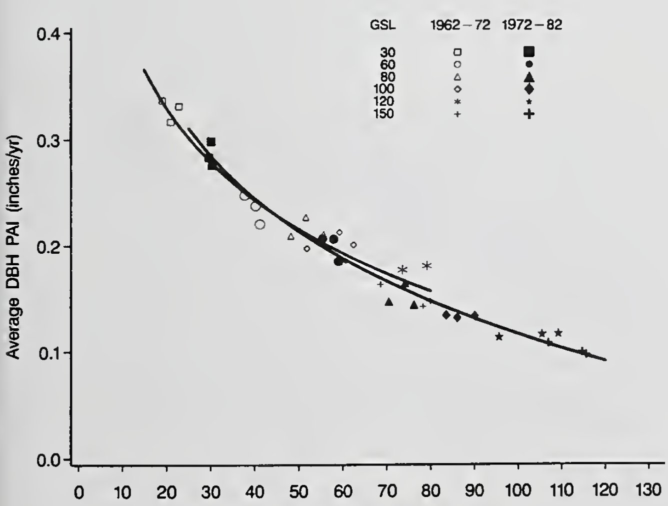

Figure 3. Relationship of periodic annual average diameter growth to residual basal area, Taylor Woods levels of growing stock study.

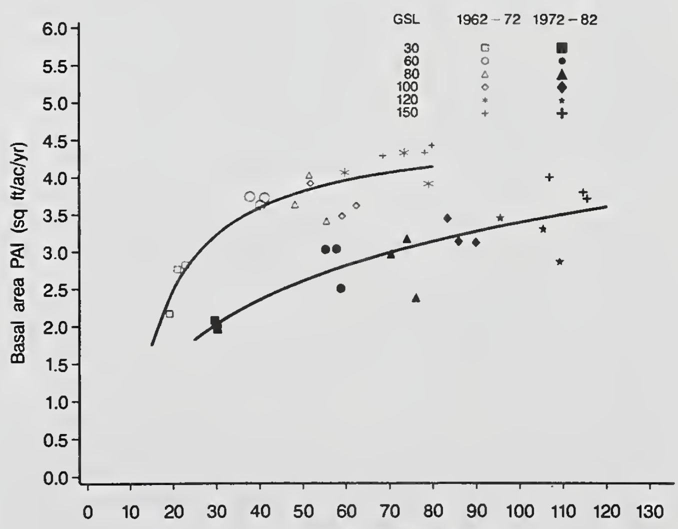

Figure 4. Relationship of periodic annual basal area increment to residual basal area, Taylor Woods levels of growing stock study.

a time period as a decade and certainly over longer cutting cycles. This commercial volume production available for harvest is a critical element in the economic feasibility of any of the proposed management regimes. Equally important is the availability of markets for the range of sizes of material produced by these periodic treatment operations. Without a viable economic return, sustained management of these stand structures will likely not be possible. Average standing merchantable volume per acre after the latest thinning in 1992 and removals are shown in figure 6. The decision to increase the residual density of the most open plots in 1992 resulted in a smaller merchantable yield from the latest thinning. If the open density had been held at 30 square feet per acre, the three lowest densities would have produced commercial yields with the last two thinnings. All treatments but the highest density produced commercial yields at the last thinning, but material removed from the denser plots was smaller in diameter.

Additional considerations in developing management strategies

As discussed above, viable markets and a commercial economic return are required to successfully develop and implement these management strategies. It is doubtful that society can or will subsidize these operations over landscape scales and in perpetuity. Successful implementation will often require treatment of small diameter material. Either markets for this material must be available or returns from harvesting larger trees will have to offset the expense of treating smaller trees.

While the goal of these management strategies is to increase and maintain structural diversity and not balanced, regulated diameter structures, adequate tree and stand projection systems are needed to refine management strategies and predict future stand conditions and possible yields from partial harvesting activities. While density and operational growth models are available for stands of ponderosa pine in the Southwest (Edminster 1988; Edminster et al. 1991), these models need refinement for use in grouped spatial arrangements in uneven-aged structures. Existing growth models are generally inadequate in projecting growth and mortality of large diameter, old trees. In addition, currently available models are not sufficiently sensitive to low residual densities proposed to maximize individual tree growth. Researchers need to collaborate with managers in monitoring responses from these treatment when applied at operational scales to improve growth projection systems.

The scale at which these treatments are applied in specific forest conditions also needs careful consideration. An underlying concept is that the treatments should be designed to work more closely with nature rather than fighting natural processes. However, the advantage may not always directly lie with following ecological theory but rather in reducing the investment in silvicultural treatments over the long term (Smith 1994). The manager needs to consider existing structure and

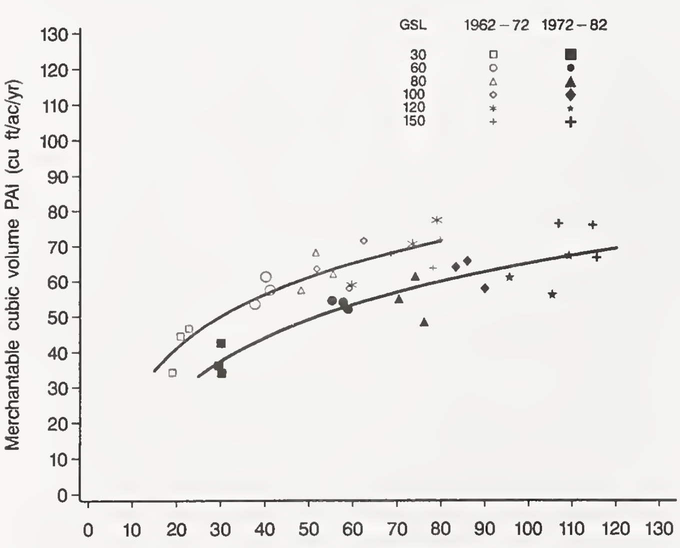

Figure 5. Relationship of periodic annual merchantable cubic volume increment to residual basal area, Taylor Woods levels of growing stock study.

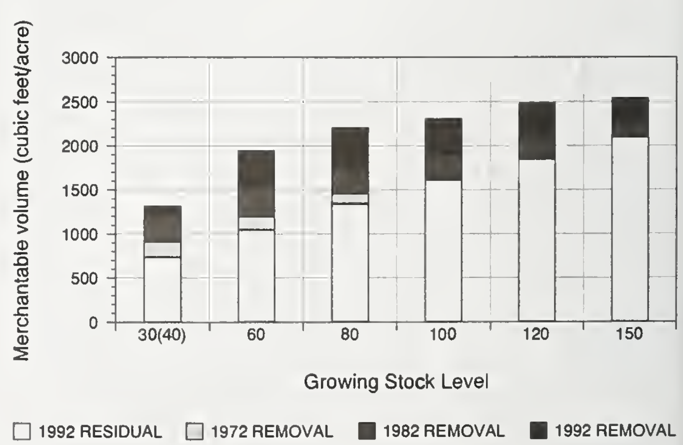

Figure 6. Standing merchantable volumes, 1992, and periodic removals by growing stock level, Taylor Woods levels of growing stock study.

how it got to be that way before embarking on an altered management strategy. In many cases, it may not be feasible to manage at a small group level, and a forest of even-aged stands of various ages and densities may adequately meet management objectives. An example of where uneven-aged management using a group selection cutting method will likely not meet management objectives is in stands infected by dwarf mistletoe (*Arceuthobium vaginatum* subsp. *cryptopodum*). Uneven-aged management has been demonstrated to not be effective in controlling dwarf mistletoe and often results in increased infection levels (Heidemann 1977). Sporadic regeneration establishment and varying growth rates will also make attaining a rigidly balanced diameter distribution a futile exercise over even large areas.

Another consideration is the dynamic nature of forest stands. As discussed above, basal area may well increase 20 to 30 square feet per acre in a decade. This relatively fast increment in stand basal area will likely require frequent treatments to control susceptibility to bark beetles (*Dendroctonus* spp.) (Oliver 1995; Schmid 1987). While hazard ratings are available for bark beetles in ponderosa pine (Sartwell and Stevens 1975; Schmid and Amman 1992; Schmid et al. 1994), most of these systems were developed for even-aged structures. The interactions of hazard ratings with density and structure are not well understood in uneven-aged conditions.

The use of silviculture as a tool for creating and maintaining stand and forest structures in southwestern ponderosa pine discussed in this paper is a relatively new undertaking. As with any shift in management focus, care is needed. A comprehensive monitoring program to assess results and guide refinements in future activities is a key to future success.

Acknowledgments

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Flexible Silvicultural and Prescribed Burning Approaches for Improving Health of Ponderosa Pine Forests

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Abstract. Prior to 1900, open stands of large, fire resistant ponderosa pine (occasionally with western larch) covered extensive areas of the West. Since the early 1900s, virtual elimination of low-intensity fires in ponderosa pine and pine/mixed conifer forests has resulted in major ecological disruptions. Today, many stands support dense thickets of small trees (often firs), and manifest insect/disease infestation and high potential for severe wildfire. These forests cover tens of millions of acres and are the focus of forest health concerns. Restoration efforts are complicated by profound changes in stand composition and structure, poor tree vigor, and fuel accumulation. Returning fire under these conditions could fatally damage already stressed overstory trees. Restoring more natural and sustainable conditions often requires a combination of silvicultural tree removal in terms of species, number, and size. In some stands, thinning from below will be sufficient, while in others, selection cutting will be needed to reduce overstory density and allow regeneration of shade-intolerant species. Depending on overstory composition and observed regeneration patterns, openings may require planting to ensure regeneration of desired species. Because fire was historically the primary disturbance agent in ponderosa pine/larch types, prescribed fire should be considered in any restoration efforts. However, most forests in need of restoration cannot be effectively treated by fire alone. Linking appropriate silvicultural and prescribed fire is the key to restoration. Prescribed fire is generally the most effective means of reducing high fire hazard, eliminating large numbers of understory trees, stimulating seral herbaceous and shrubby vegetation, creating receptive seedbed, and transforming nutrients into an available form. However, after decades of fire exclusion, existing forest conditions require a cautious but determined approach to fire application.

Introduction

For millennia, fire has shaped the structure and species composition of ponderosa pine (*Pinus ponderosa*) forests in the Inland West. Prior to the early 1900's, these forests were characterized by frequent, low- to moderate-intensity fires that killed few overstory pines. Surface fires ignited by lightning or Native Americans typically occurred at intervals of 5 to 30 years, thinning small trees and producing open, park-like, fire-resistant stands (Arno 1988).

From a species composition point of view, fire favored species like ponderosa pine that are fire-resistant and require burning to regenerate and compete successfully (Pyne 1982; Agee 1993). This shade-intolerant seral species maintained dominance on many sites in the face of strong successional pressure from more shade-tolerant — but less fire resistant — associates [e.g. Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), white fir (*Abies concolor*), and incense cedar (*Calacedrus decurrens*)]. Great longevity and exceptionally thick bark are attributes that favor pine in an environment dominated by frequent surface fires. Even after 60 to 80 years of generally successful fire suppression, pure and mixed ponderosa pine forest types still cover over 30 million acres in the western United States (Van Hooser and Keegan 1988).

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The problem

Beginning in the early twentieth century, disruption of the historic pattern of frequent fire in ponderosa pine forests brought about gradual but profound ecological changes. These changes include increased stand density and associated low tree vigor, increasingly severe insect and disease epidemics, gradually changing species composition, increased fuel accumulation, and more severe wildfires. For example, more than a million acres in eastern Oregon's Blue Mountains now consist of mostly dead or dying trees, primarily fir thickets impacted by insect and disease epidemics (Mutch et al. 1993). Epidemics of this magnitude are biological manifestations of stand conditions (i.e., density, composition, and structure) that are not sustainable in these semi-arid environments. Furthermore, large stand-destroying wildfires which were once rare in open ponderosa pine forests have become common in the dense stands that have developed as a result of fire exclusion. Dense fir thickets in the understory provide abundant fuel ladders that allow fires to increase in intensity and burn explosively through tree crowns in the overstory. Intense crown fires in ponderosa pine stands made up a large portion of the three million acres that burned in the Inland West in 1994, providing stark evidence of the vulnerable forest conditions that now prevail across the region.

A different but somewhat comparable condition exists in many unlogged, fire-excluded old-growth ponderosa pine and pine-fir-larch stands. Dense understories in these stands threaten the old-growth trees in two ways. First, intense competition for limited resources results in physiological stress. Stressed old-growth ponderosa pine are particularly vulnerable to the western pine beetle (*Dendroctonus brevicomis*) (Johnson 1972). Second, dense understories provide surface fires an avenue into the crowns of old-growth trees in the overstory.

The objective of this paper, then, is to describe silvicultural and prescribed burning techniques being tested in the restoration of second-growth and old-growth ponderosa pine stands in the Inland West.

Restoration treatments

A primary goal of restoration treatments is to create more open stand structures consistent with historic disturbance regimes, thereby increasing tree vigor and reducing vulnerability to fire and insects. A second major goal is to manipulate the existing species composition and site conditions to favor regeneration of seral ponderosa pine.

Restoration efforts are complicated by profound changes in stand composition and structure, poor tree vigor, and fuel accumulation. Returning fire under these conditions could fatally damage already stressed overstory trees. For these reasons, restoring ponderosa pine forests to more healthy and sustainable conditions will generally require a combination of silvicultural cutting and prescribed burning treatments (Fiedler et al. 1992).

A primary advantage of cutting is that it allows for the controlled removal of specific trees in terms of number, size, and location, as well as manipulation of species composition throughout all canopy strata. Cutting trees also allows them to be used for forest products, often generating enough income to pay for the needed treatments. Cutting is also an effective means of removing trees that cannot be specifically targeted and killed in a prescribed underburn. Conversely, primary advantages of prescribed burning are controlled reduction of high fire hazards, efficient elimination of large numbers of small trees (particularly firs), stimulation of herbaceous and shrubby vegetation, creation of receptive seedbeds, and transformation of nutrients from unavailable to available forms.

Three ongoing research/demonstration studies in Montana provide examples of cutting and burning treatments being used to restore stands representative of the following conditions:

1. Dense, second-growth even-aged stands,
2. Overstocked uneven-aged stands with a heavy fir component in the understory, and
3. Old-growth stands with overstocked overstories and dense, primarily fir understories.

Before silvicultural cutting and prescribed burning treatments are initiated, general restoration goals need to be established in the form of a target stand or desired future condition. Historical descriptions, photo records, and research plot data can be pieced together to provide initial targets for restoration. For example, long-term growth plot data (Barrett 1979) provide density targets for thinning or shelterwood cutting in even-aged second-growth stands. Likewise, interim results from uneven-aged stocking plots (Fiedler et al. 1988; USDI 1993) provide reasonable residual density guidelines for securing regeneration of ponderosa pine in uneven-aged stands.

For old-growth, early written accounts provide a profile of the structure and composition of stands in the ponderosa pine type. Weidman (1921), Anderson (1933), and Gruell et al. (1982) report that virgin stands in this type were primarily ponderosa pine and uneven-aged, or quoting Meyer (1934) "the typical ponderosa pine forest of the Pacific Northwest is fairly pure, fairly open, and many-aged." Anderson (1933) noted that "Few timber trees west of the Great Plains are better adapted to selective logging than ponderosa pine. Fires, insect depredations and mortality from old age through-

out the past two or three centuries have resulted in uneven-aged stands with a rather irregular distribution of age classes ranging from young seedlings to 600-year-old veterans." Tree-ring analysis and reconstruction of age-class structure in old-growth ponderosa pine stands in western Montana corroborates these earlier descriptions (Arno et al. 1995).

Collectively, these sources provide a basis for establishing density, structural, and compositional goals for currently overstocked and declining even-aged, uneven-aged, and old-growth stands. They also provide a basis for formulating restoration treatments to direct succession toward the appropriate desired future condition.

Second-growth, even-aged stands

Reasonable guidelines for post-treatment density range from 40 to 80 ft^2 of reserve basal area/ac in even-aged or irregular even-aged stands. This basal area density range is equivalent to about 50 to 100 12-in. tree per acre (tpa), or about 30 to 60 16-in. tpa. Reserve basal areas will be on the lower end of this range for drier sites and where regeneration of ponderosa pine is one of the treatment goals, and toward the upper end of the range on better sites and where the goal is thinning rather than regeneration. Somewhat lower or higher residual density goals may be appropriate for specific objectives or localized conditions.

A recent project at the Lick Creek Demonstration Area on the Bitterroot National Forest in Montana illustrates restoration efforts in a dense, 80-year-old second-growth ponderosa pine stand. The pretreatment stand had a basal area of 120 ft^2/ac and averaged 240 tpa, an overstocked condition for this relatively dry site. Basal area was reduced to 48 ft^2/ac through shelterwood cutting. The largest and highest quality pines were retained to provide shelter and a ubiquitous seed source, and because such trees were a characteristic component of pre-1900 stands. While this cutting can be described as a shelterwood, it is the first step in a long-term restoration effort to develop uneven-aged stand structures, so is really the initial cut in the implementation of the selection system. The next entry is planned in 20 to 25 years, at which time some shelter trees will be removed in a selection cutting, providing additional site resources for the remaining overstory trees, and again creating additional openings for establishment of a new age class of ponderosa pine. Future cuttings are planned at 20- to 25-year intervals indefinitely into the future, with the purpose of reducing stand density and regenerating a new age class at each entry. The long-term goal is to create and maintain a multi-aged ponderosa pine stand, allowing some of the overstory trees to reach an old-growth state.

Underburning was prescribed following the harvest activity to reduce the fire hazard from harvest-generated fuels (limbs and tops), reduce the unsightly nature of logging slash and the physical barriers to deer and elk movement, release nutrients bound in the organic horizons, and stimulate growth of herbs and shrubs.

The initial focus was on the fire hazard, which was represented in three strata of fuels. The top stratum was comprised of interconnecting crowns, which allow a wildfire to spread rapidly through the overstory. This hazard component was greatly reduced by the harvest cutting, which left 10- to 30-foot gaps between leave trees. The middle stratum — comprised of seedlings and saplings and commonly referred to as ladder fuels — was somewhat reduced by the logging operation. The third stratum, located on the ground where fires start and spread, consisted of litter, branch wood, and downed trees.

Underburning reduced the initial 4.5 tons/ac of litter and small woody fuels by nearly two-thirds. Besides the reduction of mid-stratum ladder fuels that occurred as a result of the harvest operation, an additional 60 percent of the seedlings and saplings were killed in the underburn — primarily diseased and suppressed small trees that were not wanted as part of the future stand. The average preharvest large woody fuel loading of five tons/ac was increased to seven to eight tons with logging slash. Burning returned these fuels to about their preharvest levels. These light loadings may well resemble presettlement conditions under which large fuel buildups were limited by frequent surface fires.

Prior to treatment, understory vegetation was sparse and in poor vigor because of the dense overstory. The cutting and burning treatments have resulted in improved vigor and increased flowering in the herbaceous component. Some introduced (and undesirable) herbaceous species have increased in frequency since treatment, and their long-term trends are being measured. Small increases are probably inevitable with site disturbance, but these can be minimized through use of low-impact harvesting equipment and techniques. The alternative regime — fire suppression and no management activity over the past 80 years — has led to a gradual decline in the cover and diversity of native herbs and shrubs. Eventually, this alternative regime will likely lead to a severe wildfire and a subsequent major increase in undesirable exotic species.

Overstocked, uneven-aged stands

Selection cutting in overstocked, uneven-aged stands has the multiple purpose of securing regeneration of ponderosa pine, regulating the numbers of trees throughout the diameter range and thereby increasing

the vigor of reserve trees, and reducing the percentage composition of Douglas-fir.

Using the traditional individual tree selection method, uneven-aged stand structures are typically regulated by three parameters:

1. Residual density (basal area after harvest),
2. Maximum tree size (diameter of largest trees remaining after harvest), and
3. "*q*" (ratio between numbers of trees in successive diameter classes).

Post-treatment basal areas of 40 to 60 ft²/ac are recommended for second-growth, uneven-aged stands to ensure regeneration of shade-intolerant ponderosa pine (Fiedler et al. 1988). Somewhat higher basal areas are acceptable in uneven-aged stands with a moderate old-growth component. The maximum tree diameter after harvest is commonly set at 20 to 24 in. for ponderosa pine, depending on site quality and other management objectives. A reasonable choice for *q* — the ratio between trees in successive diameter classes — is about 1.2 to 1.4 for 4-in. diameter classes (equivalent to a *q* of 1.1 to 1.2 for 2-in. classes). These low *q* values serve to allocate a relatively large proportion of the basal area to large trees.

An ongoing research/demonstration project at the Lick Creek site in Montana provides a specific example of restoration cutting and burning in overstocked, uneven-aged stand conditions. The pretreatment stand had a basal area of about 110 ft²/ac, and stem density of 160 tpa. The desired future condition of this restoration effort has a reserve basal area of 50 ft²/ac, a maximum tree diameter of 20 in. (and a minimum regulated diameter of 4 in.), and a *q* of 1.2 (4-in. classes). This target uneven-aged stand of the future has a species composition of $\geq$90 percent ponderosa pine, and about eighteen 4-in. tpa, fifteen 8-in. tpa, twelve 12-in. tpa, ten 16-in. tpa, and nine 20-in. tpa, for a total of 65 tpa.

The existing uneven-aged stand was leave-tree marked to approach as closely as possible the desired future condition in terms of density, structure, and species composition. Experience has shown that leave-tree marking results in a superior reserve stand, because the marker focuses on the highest quality trees at an appropriate spacing. With cut-tree marking, the residual stand is simply comprised of the trees that were not cut (the leftovers). Typically, the reserve basal area target can be approximately achieved in the first entry. However, because the emphasis in leave-tree marking is on tree quality and juxtaposition, only modest progress can often be made in the first entry toward the maximum diameter and diameter distribution targets.

The restoration prescriptions developed for use in uneven-aged stands are based on the same three parameters — reserve basal area, maximum diameter, and *q* — as classic individual tree selection prescriptions. However, the levels of these parameters and the implementation of restoration prescriptions are modified considerably for this application. For example, the *q* values used in restoration prescriptions are lower than those traditionally recommended in the literature. This modification serves to allocate a considerable proportion of the basal area in larger trees, which is consistent with the historic structure of stands that developed under a regime of frequent surface fires.

Another difference is that scattered small openings up to 1/4 ac in size (and occasionally larger) are created every one to several acres, on average, in the marking process. Some existing small openings are enlarged by marking trees on the periphery for leave as needed to create patches that vary in size and shape. Under this approach, each tree is evaluated, only with more stringent standards required for leave trees around openings. These openings serve to favor regeneration and early growth of seral, shade-intolerant ponderosa pine.

If a structure with more large trees is desired, a given amount of basal area — say 10 to 15 ft²/ac — can be allocated to trees larger than the maximum diameter. These trees are not included as part of the regulated diameter distribution, but are included in residual basal area calculations. This approach allows keeping some very large trees in the stand at all times to meet certain wildlife or visual objectives, or more closely emulate historic structures.

Following selection cutting at the Lick Creek site, a precommercial thinning of trees <7-in. diameter greatly reduced the density of the mid-stratum ladder fuels. An intermediate intensity understory burn was carried out during the spring following the selection cutting and associated thinning. While nearly 20 percent of the leave trees in the 4-in. diameter class had died from burning injury during the first two years after treatment, only 3 percent had died in the 8-, 12-, and 16-in. classes, and none in the 20-in. class. The 20 percent greater number of crop trees left in the 4-in. class initially (resulting from a q of 1.2) helped offset the higher mortality rates that were anticipated in the smaller diameter classes. Furthermore, the prescribed burning was successful at the operational level in terms of accomplishing silvicultural and ecological goals. Seedling-size Douglas-fir were virtually eliminated, fire hazard from logging slash was greatly reduced, forest floor materials were recycled, and seedbed preparation was accomplished over most of the area.

Overstocked, old-growth stands

Because site utilization as measured by Stand Density Index (SDI; Reineke 1933) is less per square

foot of basal area in large trees than in small trees (Fiedler and Cully 1994), somewhat higher reserve densities can be maintained in old-growth than in second-growth stands. For example, the SDI associated with a square foot of basal area in 24-in. trees is only about half that of a square foot of basal area in 4-in. trees. It follows that stands with a high proportion of their basal area in large trees will provide a lesser draw on site resources that stands with the same total basal area, but with a greater proportion of their basal area in small trees.

An ongoing project at the University of Montana's Lubrecht Experimental Forest illustrates restoration activities in an overstocked, old-growth ponderosa pine stand. The pretreatment stand had a basal area of about 130 ft^2/ac, which was approximately evenly divided between an old-growth ponderosa pine/Douglas-fir overstory stratum, and sapling, pole- and small sawtimber-sized fir and pine comprising several lower- and mid-canopy strata. A modified individual tree selection cutting (methods discussed previously) was carried out which reduced the overall stand basal area to about 65 ft^2/ac, and strongly favored ponderosa pine for retention. Numerous small openings were created in the marking process to favor subsequent regeneration of ponderosa pine. In contrast to the common approach toward marking to increase the uniformity of spacing, occasional groups of old-growth trees were left intact (or nearly so) to maintain the inherently clumpy nature of these stands.

Prescribed burning in old-growth stands requires a cautious approach, particularly in the first application, because of decades of fuel accumulation. Of even greater initial concern is the dense understory of seedlings and saplings that commonly occurs in these stands. Ladder fuels (especially fir) can readily ignite in an underburn, and either scorch the already thin crowns of old-growth trees or allow fire to move into the overstory. Dense understories are best thinned first, then piled and burned before applying a broadcast underburn. While fuel accumulation is a general problem, it is most acute around the bases of old-growth trees, where mounds of needles and sloughed bark a foot deep or more may have accumulated over the years. Either a relatively low intensity fire should be prescribed for the first underburning after decades of fuel buildup, or the loss of a considerable number of old-growth trees from root and cambial damage can be expected.

Restoration challenges

Social

Private citizens, environmental organizations, and wildlife biologists, as examples, may not support harvesting trees in overstocked stands in general, and old-growth stands in particular. While forest health and increasing risk of forest fires has been widely covered in both the popular media and professional forums, many people are still skeptical. Critics question the true motivations behind proposed treatments. Furthermore, dense stands with dead and dying trees may be valuable habitat for some wildlife species, with any change resulting from cutting or prescribed burning perceived as negative.

Technical

High stand density, structures that include trees of all sizes, heavy fuel accumulations in the form of dead and down material, and thick mounds of sloughed bark and needles around the base of old-growth trees all present special challenges to the reintroduction of fire. Harrington and Sackett (1992) note that attempting to reduce decades of fuel buildup in a single burn could result in the loss of 20 to 50 percent of the old-growth trees in a stand. They also report that while it is technically feasible to physically remove heavy fuels from around old-growth clumps before burning, this alternative is limited by cost.

Future implications

Restoration treatments cannot be considered a one-time activity — they must be integrated within a long-term plan for future treatments. More specifically, existing stand conditions, prescribed treatments, and post-treatment conditions should be documented to provide a benchmark against which future conditions can be compared, and progress toward the desired (target) condition can be measured.

Application of the proposed restoration treatments should provide a significant first step toward achieving the appropriate desired future condition. Judicious application should significantly improve stand vigor, reduce vulnerability to insects, disease, and severe wildfire, and increase management options in the future.

Alternatively, the scenario of the future is indeed a troubling one if restoration activities are not undertaken. The effects of disrupted ecological processes will become even more severe. Insect and disease epidemics will likely increase in extent and severity, and the conversion from low-intensity to stand-replacement fire regimes will accelerate. Furthermore, the demise of old-growth stands and remnants will continue, coupled with the equally disturbing fact that few replacement stands are developing because of ever-increasing stand densities.

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Mt. Trumbull Ecosystem Restoration Project

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Abstract. This paper describes a pilot project to provide ecological information about changes in ponderosa pine and a "learn by doing" operational-scale adaptive ecosystem restoration experiment. This project would be implemented in the next few months in the Mt. Trumbull area in northwest Arizona on BLM administered land. This land provides important wilderness, wildlife habitat, and recreational opportunities.

In recent years, the number and size of wildfires throughout the United States has increased. Costs of these fires in terms of lost lives, private property and natural resources, as well as fire fighting costs, have significantly increased. This proposed project directly addresses this problem and thus has widespread value. Research indicates tree population irruptions resulting from fire exclusion have led to declining trends in both plant and animal diversity, vegetative productivity, tree vigor, and water availability. At the same time, fuel loads, disease and insect epidemics, and crownfire severity have increased creating unhealthy ecological conditions. Reversing these trends is clearly desirable.

A comprehensive package of mechanical, biological, and pyric treatments coupled with scientific research is proposed. In contrast to historical forest management in which old-growth trees are removed and younger trees are left behind, our restoration treatments would consist of removing younger trees and leaving the large, yellow-barked pines. Heavy forest floor accumulations would be raked away from old-growth trees; then smaller tree removal and prescribed burning would occur. Once restoration treatments are in place, a cost-effective prescribed natural fire regime would be implemented. The proposed ponderosa pine restoration area would exceed 10,000 treated acres. Some of these acres are inside designated wilderness, which poses specific challenges and restrictions. Simultaneous management research and evaluation would utilize university scientists. Collaboration among agencies, interested publics, and the scientific community, and adaptive management are integral parts of this proposal.

Introduction

In recent years the number and size of wildfires has increased. The costs of these fires in terms of lost lives, private property and natural resources, as well as dollar costs of fighting the fires, have also significantly increased. This project directly addresses this problem and thus has widespread value. A detailed study plan will be developed once initial project plans are completed.

The benefits from ecosystem restoration in the ponderosa pine type will accrue to both local and national constituencies. Research indicates that tree population irruptions resulting from fire exclusion have led to declines in both plant and animal diversity, vegetative productivity, tree vigor (especially old-growth trees), and on- and off-site water availability. At the same time fuel loads, disease and insect epidemics, and crownfire severity and size have increased at exponential rates creating unhealthy ecological conditions. Reversing these trends is clearly desirable. This project could serve as a prototype for ponderosa ecosystem restoration.

The proposed project lies within the ponderosa pine type of the Mt. Trumbull Resource Conservation Area in the Arizona Strip District. The project will provide information necessary for answering the question, "Can ecosystem restoration alleviate the negative environmental consequences of fire regime disruption and

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restore ecosystem health, and do so in an economically, socially, and politically cost-effective manner?". Collaboration among agencies, interested public and the scientific community is an integral part of this proposal. The experience gained and documents produced from this project will be useful to others throughout the West who are concerned with restoration of ecosystem health.

To accomplish this goal we propose a comprehensive package of ecosystem restoration treatments coupled with scientific research. The proposed project was developed by an interdisciplinary process which is described in the "Planning Approach" section below. Approximately 1,000 acres per year for ten years would be restored. The proposed ponderosa pine restoration area would exceed 10,000 treated acres, making this the largest ecosystem restoration project of its type. Some of these acres are inside designated wilderness, which poses specific challenges and restrictions. Such a large-scale project is needed to clearly demonstrate results over a management-sized area. Simultaneous management research and evaluation would be interdisciplinary utilizing university scientists with demonstrated expertise in restoration ecology, plant and animal conservation biology, ecological economics, community ecology, and fire. These scientists have a history of successful collaboration on related, but smaller-scale pine ecosystem restoration projects. They propose to document the ecological effect of the restoration treatments with replicated study plots contrasted to each other and to control plots and to produce both scholarly and public information syntheses of ecological restoration knowledge in a variety of media formats.

In contrast to historical forest management treatments in which old-growth trees are removed and younger trees are left behind, our restoration treatments consist of removing the trees which have come in since fire regime disruption (*i.e.*, the portion of the tree populations which represent an unnatural population irruption) leaving behind the old-growth trees and sufficient "post-settlement" trees to restore pre-settlement forest structure. Heavy forest floor accumulations will then be raked away from the base of the old-growth trees. Finally, prescribed burning will be used to simulate the natural fire regime of the area. Once restoration treatments are in place, a cost-effective prescribed natural fire regime will be implemented.

Intensive gathering and analysis of existing information and new data from the project would be used to produce both public education and information for a variety of media formats (*e.g.*, slide presentations, videos, radio and television, brochures, booklets, and on-site information signs and nature trails), and to fulfill the NEPA process.

Results from this adaptive ecosystem restoration project could extend far beyond the Mount Trumbull Resource Conservation Area. A clearly documented example of a practical ecosystem restoration application will be of great value to both the general public and land managers who are confronting similar problems throughout the natural range of western long-needled pine ecosystems (50,000,000 acres at the time of Euro-American settlement) and in other forest types which evolved under the influence of frequent fire.

Description of the area

The 120,000 acre Mt. Trumbull Resource Conservation Area (RCA) is located in northwest Arizona about 60 miles from the nearest paved road and is bounded on the south by the northern boundaries of Grand Canyon National Park and Lake Mead National Recreational Area. Mount Trumbull, highest point on the Arizona Strip, dominates the surrounding landscape. The RCA consists of 88 percent BLM administered public land, with the remaining area being 6 percent State land and 6 percent private land. In 1984, 22,500 acres of BLM land in what is now the RCA around Mt. Trumbull and Mt. Logan were designated by Congress as wilderness.

Ponderosa pine occurs on 13,200 acres within the RCA. Management of the ponderosa pine ecosystem is significantly affected by both past harvest and fire history and wilderness designations. Livestock grazing has also had some effects. Future pine ecosystem management in wilderness may be tied to non-motorized technologies and minimum tool consideration. Some of the ponderosa pine, chiefly that at the higher elevations and more inaccessible locations, has never been logged. Various combinations of wilderness area/never logged, wilderness area/previously logged, and non-wilderness area/previously logged exist in the ponderosa pine type.

The RCA is located nearly centered between Lake Powell and Lake Mead on an east/west axis, and between Zion National Park to the north and the Grand Canyon to the south. The Kaibab Plateau and the Kaibab National Forest lie less than thirty miles to the east. In contrast to the Kaibab Plateau, the Trumbull area does not have a recent history of intensive timber harvest. There are no existing timber sales or plans for extensive timber cutting.

Unlike most of the rest of the West, the area appears to have a relative abundance of 300-800 year-old trees. Yet overall, fire disruption has set into motion changes in ecosystem structure and function that threaten not only the survival of the old-growth trees, but also the sustainability of the entire ecosystem. Biodiversity and fire dependent species are currently declining. Recent research by Northern Arizona University scientists from small-scale studies near Flagstaff indicates that treatments which restore pre-settlement structure and

function in ponderosa pine ecosystems dramatically improve the health of old-growth trees and enhance ecosystem health and human habitat values.

Land use planning

The draft 1989 Arizona Strip Resource Management Plan (RMP) generated public comments and controversy regarding potential management of ponderosa pine and other resources in the Mt. Trumbull area. BLM responded to these public concerns in the final RMP (1992) by designating the area as the Mt. Trumbull RCA and by directing that the ponderosa forest be managed only for the enhancement of other resources, such as wilderness, wildlife, and recreation. The need for additional planning was recognized and a commitment was made to develop tailored management prescriptions for the Mt. Trumbull RCA and to involve interested publics and other agencies in future planning. Additionally, the RMP states that the area is to be managed to maintain a healthy, biologically diverse, ecosystem.

Planning approach

The Mt. Trumbull RCA planning effort offers a unique ecosystem management study opportunity. Recreational use of the area is closely tied to adjacent Park Service lands, as well as state and private lands within the boundaries. An interdisciplinary planning team consisting of BLM, AGFD, NPS, local citizens, livestock grazing permittees, Kaibab Industries Inc., and conservation organizations was invited to develop this plan. A mission statement was developed stating that by consensus the team would define future conditions for the Mt. Trumbull area while maintaining sound ecosystems. A flexible boundary for the study area was established which can be adjusted as the plan is implemented. Through consensus, the interdisciplinary planning team first identified issues and then developed draft management objectives to resolve these issues.

Adaptive management

Using the best available science, an adaptive ecosystem management approach is proposed to meet objectives and carry out proposed management. Since the RCA plan is adaptive it is expected to have an indefinite life span. Potential actions would be limited by tool availability, constraints on tool use, and available funds. Potential actions would be considered by the Mt. Trumbull RCA Team through a screening process and subject to a site specific NEPA analysis. Subsequent decisions would be made by monitoring and evaluating the planned management actions and their effectiveness. Plan refinement and monitoring would be an ongoing process by the team, interested public, and ad hoc interagency management and research peer reviews. We anticipate that the approach being tried at Mt. Trumbull to solve existing ecosystem problems will be successful and could serve as the foundation for developing ecosystem restoration and management programs throughout the ponderosa pine type.

Specific planned actions

A brief description of the nine planned specific projects or actions follows:

1. ***NAU-guided scientific studies and public education plots.*** An integrated set of research and demonstration studies will quantify the effects of restoration treatments on vegetation and fuel structure, wildlife and human habitat characteristics, aesthetics, old-growth tree vigor, and biodiversity of insects which represent critical components to ecosystem structure and function (food webs, plant population controllers, nutrient cycling, *etc.*). These studies consist of a combination of historical and experimental studies. First, to estimate the long-term effects of different treatment scenarios (including no action), dendroecological studies of existing crownfire, bark beetle kill, thinning, and prescribed burn sites in the Mount Trumbull area will be conducted. Second, controlled experimental procedures will be used to implement management treatments including combinations of mechanical thinning of post-settlement trees, induced bark beetle thinning (using pheromone baits), and prescribed burning. Five sets of replicated 20-acre restoration treatments would be established to test and demonstrate mechanical and prescribed burning ecosystem restoration treatments. Treatments would consist of control, thin only, thin and burn, and burn only applications. A companion set of 30 one-acre spot treatment test plots would be established to examine mechanical and bark beetle thinning spot treatments for potential use in wilderness restoration. These experimental plots would be distributed primarily in the areas described in items 3 and 4 below. Operational projects would be designed to accommodate the experimental studies.

   A derived benefit from these management treatments will be the development of a model system to evaluate several key ecological indicators and processes as measures of resource condition. Biological indicators of ecosystem health allow for the continued

monitoring of resource condition following natural or anthropogenic disturbance and stress. The development of biological indicators of ecosystem health would have broad applicability to pine forests of the western U.S.

2. ***Public information, education, and involvement.*** Public outreach efforts to inform the visiting public and help them understand and participate in the restoration efforts would include the design and construction of a small, centralized contact station in the Nixon Flat area. This station would provide information and interpretation covering diverse local resource topics around the central theme of ecosystem restoration. Additionally, one of the accessible, easily visible, project sites would be selected as a public demonstration site and would provide on-site interpretation of the restoration efforts.

   In addition to reaching the visiting public on-site in the Mt. Trumbull area, off-site information and interpretation efforts would be needed to "bring the public along" on the restoration efforts. Outreach would include news releases, progress updates, the design and printing of a four-color brochure, videotapes, slide presentations, and the design and fabrication of an exhibit to be displayed at the new Interagency Office and Information Center in St. George, Utah. Mobile traveling displays for presentation at fairs, schools, or civic organization functions would be developed.

3. ***Intensive ecosystem restoration research and demonstration area.*** The unit comprises 448 acres, characterized by approximately five old-growth ponderosa pine trees per acre and an abundance of younger age class trees. Many of the old-growth trees appear to be in excess of 500 years old. Management practices have led to an ecosystem which is not sustainable and which is showing increased mortality of old-growth trees likely due to competition with overstocked younger age class trees. The restoration treatment for this area will consist of removing the majority of the post-settlement trees, raking heavy fuels from the base of the old-growth trees, and burning at a frequency which approximates the natural fire regime. Fire control lines would be constructed and used in conjunction with existing roads to contain prescribed fires.

4. ***Ecosystem restoration crownfire breaks.*** Nearly 100 years of fire suppression have led to a significant build up of fuels in the RCA. Without restoration, current fuel conditions are such that continued ecosystem degradation leading to major crownfires is inevitable. Until the built-up fuels are reduced to the point where wildfires can be allowed to burn naturally, expensive fire control and suppression will continue to be needed. Therefore, initial work to restore pre-settlement ecosystem conditions will focus upon fragmenting the largest blocks of heavy fuel in order to reduce the risk of large wildfire occurrence and suppression costs if one does occur. Strips of restored forest approximately 100 yards wide and a total of 6 miles long will be created. These restored strips will serve as starting lines for subsequent restoration efforts.

5. ***Mt. Logan wilderness restoration.*** During the mid 1970s while the area was under USDA Forest Service administration, about 700 acres of the ponderosa pine type in the Mt. Logan area was mechanically thinned to reduce competition between trees and to allow faster growth of the remaining trees. The cut trees were left where they fell, causing a dangerous build up of litter. These fallen trees, along with accumulations of needles and other litter, have created a high fire risk. In about 150 acres with heavier litter, the cut trees have been piled and burned in preparation for an underburn to further reduce fire hazard. In 1984, the Mt. Logan Wilderness Area was designated and included 600 acres of the 700 acre thinning. To enhance wilderness values and reintroduce fire into the area, completion of the underburn is necessary. Old-growth ponderosa will be protected during the burn, as described in 1 above.

6. ***Mt. Trumbull wilderness spot restoration treatments.*** The Mt. Trumbull Wilderness Area contains 1,300 acres of the ponderosa pine vegetation type. Due to limited accessibility, this area has not been logged and contains a density of approximately 9-10 old-growth trees per acre. After assessment of initial spot restoration treatment effects (see number 1, above), old-growth trees will be protected from wildfire by raking forest litter away from their bases and released from competition by mechanically removing (utilizing minimum tool policy) younger age class trees within 50-100 feet from their bases. Slash will be burned under prescription. These actions, while preserving old-growth trees initially, will also allow for future use of prescribed fire, whether human or lightning ignited, with the ultimate goal of having a natural fire regime returned to this wilderness area. This area will serve as an excellent natural laboratory for testing adaptive ecosystem restoration in wilderness areas.

7. ***Ecosystem restoration and management work center.*** Public use of the Mt. Trumbull RCA is rapidly changing. BLM's use of the present Mt. Trumbull Administrative Site is also changing and the location is now used by up to 40 people at a time. Current uses include: cultural, volunteer, and univer-

sity groups, land use planning meetings, wildlife survey crews, *etc.* Furthermore, this project will require additional facilities dedicated to the ecosystem management research effort. The existing fire guard station dates back to the late 1950s and pays scant attention to esthetics. It can accommodate only eight comfortably, and has inadequate facilities for meetings, meal preparation, and restrooms to support current and projected uses. The sleeping facilities, showers, and restrooms are not adequate for accommodation of both males and females concurrently. A new 3,000 square foot building is proposed with room for meetings, a ramada for eating and sleeping under during inclement weather, restrooms, and an information kiosk for visitors in the area, plus reasonable associated provisions for fencing, signs, water, and sewer.

8. ***Planning/nepa.*** An Environmental Assessment (EA) will be prepared for the Mt. Trumbull RCA management plan. Adaptive ecosystem management will require adjustments to the proposed RCA plan as new information becomes available, which in turn will require additional NEPA compliance. Supplemental environmental assessments (EA) will be prepared for proposals which are not adequately analyzed in the original EA. Site specific cultural and/or threatened and endangered species or candidate species clearances will also be a component of supplemental EAs, as will wilderness considerations. Public review and comment will be included and will be critical for continued public support.

9. ***Administration, monitoring, evaluation.*** To effectively manage and coordinate all facets of the RCA implementation and ecosystem restoration, a project administrator and additional staff will be needed. Even with extensive use of contracts, additional staff will be needed to plan and design projects, prepare and supervise contacts, and ensure quality control. Intensive monitoring and evaluation of planned actions to ensure that objectives are being met would also be required. Agency on-the-ground project work being done concurrently with academic research, along with the required public involvement, will require considerable communication to ensure a fully coordinated and a truly adaptive ecosystem management approach. There will also be a considerable workload in disseminating the information accrued.

Paying for the project

An final objective of this effort is to determine the costs and feasibility of ecosystem restoration and public support for various types of treatments, particularly inside wilderness. It is believed that volunteers could be used to help save old-growth trees, particularly in wilderness and efforts are underway to secure such help. Wood sales and salvage of materials could off-set the cost in areas where mechanical thinning is done.

For example, in areas outside wilderness where post-settlement trees in the 5-16 inch class are mechanically removed, 3,000 - 5,000 board feet per acre suitable for industrial processing could be salvaged. The Pearson Natural Area restoration treatment, near Flagstaff, AZ, (funded by the National Science Foundation) and similar to the Mt. Trumbull area, yielded 5,300 board feet per acre from post-settlement tree removal.

Concluding remarks

As mentioned earlier, we are proceeding with prescribed burning in the Mt. Logan area to reduce existing heavy levels of fuel. This work is expected to continue and the other efforts described above will be implemented as funding is secured. Completion of a draft management plan for the Mt. Trumbull RCA and completion of the related required NEPA documentation, with full public and agency review, is the next major step.

I think you will be hearing positive things about the Mt. Trumbull project and its applications as we start to do more work on the ground.

Conservation of pine-oak forests in northern Mexico

Peter Z. Fulé[1] and Wallace Covington[2]

Abstract. Highly diverse forests dominated by numerous pine and oak species cover the upper elevations of the Sierra Madre Occidental in northern Mexico. Disturbance regimes characterized by frequent, low-intensity fires have been maintained to the present, or have only recently been disrupted, in many areas. As a consequence of continued frequent fire, the relatively open stand structures and low fuel levels of these Madrean ecosystems have been preserved, in contrast to the development of dense forests with high fuel loading following fire exclusion elsewhere in North America. Although rapid changes due to economic and population pressures in the northern Mexican mountains threaten the diversity and integrity of pine-oak ecosystems, the opportunity exists to take advantage of the fire disturbance regime in managing for long-term conservation of natural forests. Furthermore, the Mexican forests provide present-day examples of fire-adapted ecosystems which may prove highly valuable for guiding the restoration of related ecosystems degraded due to extended fire exclusion in the US and Canada.

Introduction

Striking changes in long-needled pine ecosystems following disruption of the long-term frequent fire regime have been extensively documented in the western United States (e.g., Swetnam and Baisan 1994, Covington et al. 1994, Arno et al. 1995). These changes include increased tree density, fuel loading, and erosion, together with reduced species diversity and herbaceous production, and an alarming increase in the severity and size of wildfires (Swetnam 1990, Everett et al. 1994, Covington et al. 1994). In contrast, although forests of northern Mexico are composed of fire-adapted species closely related to those further north, and thousands of fires burn each year in the Mexican mountains, high-intensity crownfires remain rare (González-Cabán and Sandberg 1989, SARH 1994). Why does the ecosystem which extends seamlessly across political boundaries exhibit such different characteristics north and south of the border?

Fire plays an important ecological role in the forests of northern Mexico. Within these diverse but little-studied ecosystems are areas where frequent, low-intensity fires have continued up to the present, or where fire regime disruption has only recently begun. These forests have great potential for long-term conservation of ecological processes, such as fire disturbance regimes, together with ecosystem structures. Furthermore, the Mexican forests can serve as one benchmark for the ecological restoration of related fire-adapted forests elsewhere in North America.

Our goal in this paper is to outline initial studies in fire ecology in the Sierra Madre Occidental, the cordilleran mountain range of western and northern Mexico, beginning by describing the diverse forests which cover the upper elevations of the Sierra Madre Occidental and the general historical and cultural setting in which humans have interacted in these ecosystems. Then we focus on a specific study in the central Sierra Madre, in which we have investigated long-term fire disturbance regimes, ecosystem structures such as forest density and dead biomass loading, and the relationships between changes in fire regimes and ecosystem structures, showing that fire is a keystone ecological process which shapes and regulates forests. Finally, we discuss the implications of fire's ecological role, implications for the conservation, management, and restoration of fire-adapted ecosystems both in Mexico and elsewhere in North America.

Biogeography

Ecosystems of closely-related, long-needled pines dominate the western cordillera of North America in a

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nearly continuous band from southern Canada to northern Mexico. Members of the taxonomic section Ponderosae, important species include ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*P. jeffreyi*), Durango pine (*P. durangensis*), Arizona pine (*P. arizonica*), and Apache pine (*P. engelmannii*) (Perry 1991). Beyond sharing a common evolutionary lineage, these species form an ecological group well-adapted to disturbance regimes of frequent, low-intensity fires (McCune 1988). Adaptations such as thick, insulating bark, protected growing buds, and the production of highly flammable litter have permitted these species to thrive under the dry, windy conditions and frequent lightning characteristic of their mountainous habitat (Mutch 1970, McCune 1988).

In northern Mexico, long-needled pine forests follow the upper elevations of the Sierra Madre Occidental (Martínez 1948, Perry 1991). This mountain range begins in the "sky islands", or Madrean archipelago, of isolated peaks in southern Arizona, then consolidates into a broad, high landmass dissected by steep canyons and river valleys through the states of Chihuahua and Durango, merging into the central Mexican highlands further south. On the western slopes of the range, down to the Pacific coastal plain in Sonora, Sinaloa, and Nayarit, dams provide water for intensive irrigated agriculture. In the rain shadow to the east lie dry foothills and desert. There are no large cities in the Sierra, but the resources of the mountain range, primarily water but also mining, timber, grazing, and agricultural production, are essential to the economy of northern Mexico. Considered over evolutionary time, the mountain range has been a corridor to connect temperate and tropical ecosystems of North America, serving as a migratory pathway and a center of endemism (Perry 1991, Manuel and DeJesús 1993, Felger and Wilson 1995).

Despite many similarities, the long-needled pine forests of northern Mexico are also distinct from the related ecosystems further north. First, a wealth of biological diversity exists in the Madrean forests (Felger and Wilson 1995). Combining the tropical influences of the southern latitude with the great elevational gradients of the canyons, highly diverse ecosystems flourish in the Sierra Madre. Twenty or more overstory tree species may occur in an area of a few hectares: seven or eight species of pines, a similar number of oaks, and other trees including alder (*Alnus* sp.), aspen (*Populus* sp.), junipers (*Juniperus* spp.), and madrones (*Arbutus* spp.). In this respect, the Sierra Madre is a microcosm of Mexico, one of the three most biologically diverse nations with approximately half the world's species in the genera *Pinus* and *Quercus* (Perry 1991, Nixon 1993, Felger and Wilson 1995). Taken as a whole, plant communities of the Madrean pine-oak forests are considered comparable in species richness to tropical rainforests and also contain numerous plants with valuable medicinal properties (Felger and Wilson 1995).

A second important distinction between Madrean forests and those further north is the persistence of frequent, low-intensity fire regimes. While lightning has continued on both sides of the border, the cultural context in which humans interact with ecosystems has been quite different in northern Mexico than in the US and Canada.

Historical and cultural context

Indigenous peoples have lived in the Sierra Madre Occidental for thousands of years. The two prominent groups of the forested highlands were the Tarahumara (Rarámuri) people, of present-day Chihuahua, and the Tepehuan people further south (Pennington 1963, 1969, Gerhard 1982). Among the adaptations these people made to life in a fire-prone ecosystem was the frequent application of fire. Traditional uses of fire included clearing of agricultural plots, flushing animals from cover, and warfare (Pennington 1963, 1969).

Spanish influence came early to the Sierra Madre because of its famous mineral riches: gold, silver, and copper. Spanish towns and mines were established as long as 400 years ago. Except where minerals were discovered, the Spanish were not interested in colonizing the inhospitable Sierra. European diseases such as smallpox, however, spread more rapidly than armies among the indigenous populations. The indigenous population of north-central Durango, for example, declined by 95% in the first two centuries following contact with Europeans (Gerhard 1982). The less accessible areas of the Sierra Madre remained sparsely populated well into the twentieth century. Large tracts were finally settled as *ejidos*, communally-held land grants.

Spanish and Mexican settlers continued to apply fire in the Madrean forests, using fire to clear fields and spur herbaceous growth. These fires generally spread into the woods, in addition to fires started by lightning, cooking and campfires, and cigarettes (see González-Caban and Sandberg 1989 for discussion of human-caused fires in Mexico). Until recently, little attention has been paid to fire suppression, probably for a combination of reasons. From the perspective of miners, farmers, and ranchers, the effects of fire in reducing tree density and fertilizing herbaceous growth are highly positive. The trees themselves were often not especially valued, since little infrastructure existed for timber extraction, and in any event neither the government nor the local populations had the economic resources to train, equip, and pay wildland firefighters. A similar lack of infrastructure and investment kept the mountains

(though not the valleys) of northern Mexico free from the immense herds of cattle and sheep which were introduced to Arizona and New Mexico following the railroads (Leopold 1937, Cooper 1960, Covington and Moore 1994).

The traditional Mexican attitude toward wildfire contrasted sharply with the concerns of Anglo-American settlers in the western United States, especially after the establishment of the Forest Service. Under Gifford Pinchot and his successors, the new agency decried the devastating effects of wildfire and seized upon fire control as a defining symbol of the forestry profession, applying extensive government resources to the task (Pyne 1982).

The differences between Mexican and American policies were apparent early on. In 1937 Aldo Leopold contrasted the open forests, lush grasslands, and flowing streams of the Chihuahua mountains with the dense trees, heavy fuels, and dry, eroded soils of the southwestern US, commenting that "[t]he Chihuahua Sierras burn over every few years. There are no ill effects...But the watersheds are intact, whereas our own watersheds, sedulously protected from fire, but mercilessly grazed before the forests were created, and much too hard since, are a wreck." Twenty-five years later, Joe Marshall (1962) reinforced Leopold's observations, saying:

> "If we grant that this contrast is too abrupt and too coincident with the international boundary to be due exclusively to climate, then we must look for man-made causes of the difference in vegetation. The outstanding cause seems to be fire... There has until very recently been no protection on the Mexico side from naturally occurring fires. These sweep through the grass, killing young junipers, eliminating debris, and leaving unscathed the tall clear trunks of the widely spaced pines."

Although ecologists have commented on the importance of fire in forested ecosystems of northern Mexico for some time, relatively few studies have been done to quantify the forest structures and disturbance regimes of northern Mexico (Gonzáles-Cabán and Sandberg 1989, Rodríguez-Trejo and Sierra-Pineda 1992). Dieterich (1983) found that frequent fires, with a mean fire return interval (MFI) around 4 years, continued up the 1970's in the Sierra de los Ajos of Sonora, just south of the Arizona border. More recently, this cross-border comparison has been expanded (Baisan and Swetnam 1994). In a different vegetation type, Richard Minnich (1983) compared chaparral ecosystems in southern California and northern Baja California, showing that fire exclusion on the US side had created vast contiguous fuelbeds of mature, flammable chaparral, in contrast to the mosaic of fuel patches maintained by frequent small fires in Mexico. But little is known about fire ecology south of the area immediately adjacent to the US border.

In 1993 we initiated a study in northern Durango to investigate the frequency of fire and its effects on forest structure in the central Sierra Madre Occidental. We were interested in looking at an ecosystem in which the long-term patterns of disturbance were intact and where forest structure had been relatively unaffected by harvesting. Our specific goals were (1) to determine the historic fire frequencies at a series of sites, (2) to compare ecosystem structures under different fire regimes, (3) to examine the effects on forest density, regeneration, and fuel loading, and (4) to relate the findings to long-term strategies of conservation or restoration of similar fire-adapted forests in Mexico and elsewhere in North America. Initial findings from this study were reported in Fulé and Covington (1994).

Study sites

In 1993 and 1994 we established four study sites in northwestern Durango at the crest of the Sierra Madre Occidental, centered around latitude 25° 03', longitude 106° 13'. Three of the sites were in unharvested pine-oak forest, the fourth in a mixed conifer forest with true fir (*Abies durangensis*) and Douglas-fir (*Pseudotsuga* sp.). One location, near Topia, represented the western edge of the Madrean forest, while the other three were grouped near the center of the range. Study site elevations ranged from 2200 m (7200 ft) to 2500 m (8200 ft) for the pine-oak sites, up to 2960 m (9700 ft) at the mixed conifer site. Large-scale timber harvesting in this region of Durango was begun only in the 1970's, but currently few areas remain unharvested. Soils are of igneous origin, mainly rhyolitic with scattered basaltic and granitic outcrops. Weather records are limited to the last decade; annual precipitation at two regional weather stations ranged from 1540 mm (61 in) to 2200 mm (87 in), with most rain falling in the summer monsoon season. Throughout the region, forest uses include grazing of domestic animals, dispersed tree cutting for building logs, shingles, and firewood, and clearing of small, remote parcels for illicit cultivation. For further information on site characteristics and sampling methods, see Fulé and Covington (1994).

At each site we established sampling plots on a grid to systematically measure the overstory, understory, fuels, and fire disturbance history. We chose these methods to link together ecosystem structures such as tree density and age distribution with the frequency of fire. On each circular plot we recorded the species, condition, diameter at breast height (dbh), crown ratio, and presence of lightning scars on each tree. Increment cores were collected from all conifers over 6 cm dbh to determine age. Understory and herbaceous density

were tallied on nested subplots and fuels were measured along a planar transect (Brown 1974).

To determine each site's fire history, fire scar samples were collected from catfaces on living and dead trees near each sample grid point. A complete or partial cross-section was taken from each sample tree. The cross sections were cut flat and sanded with increasingly finer abrasives up to 400 grit sandpaper. Tree-ring patterns were then crossdated with master tree-ring chronologies developed in this and other studies (Stokes and Smiley 1968, Harlan 1973). Tree-ring widths were measured and crossdating was checked with the COFECHA program (Grissino-Mayer and Holmes 1993) and fire histories were analyzed with the FHX2 software[3] (Grissino-Mayer 1995).

[3] *The use of trade names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product.*

Fire regimes

Two common themes emerged from the fire history analyses. First, fires were frequent, from as far back as the records extended, up through the mid-twentieth century. Second, an extended period of recent fire regime disruption was apparent at three of the four study sites. The fire history of the Ejido Topia site (approximately 30 ha [74 ac] in size) is an example of these trends (Figure 1). Fires were recorded over a 229 year period from 1764 to 1993. As the record goes further into the past, fewer trees record fires because trees die and snags and logs are consumed in the recurring fires. At the Ejido Topia site, a good record is available from 1812 to the present. Fires recurred regularly from 1812 to 1955, with a mean fire return interval (MFI) of 4.1 years (min. 1 year, max. 9 years, Weibull median probability interval [WMPI] 4.0 years) for all fires, and

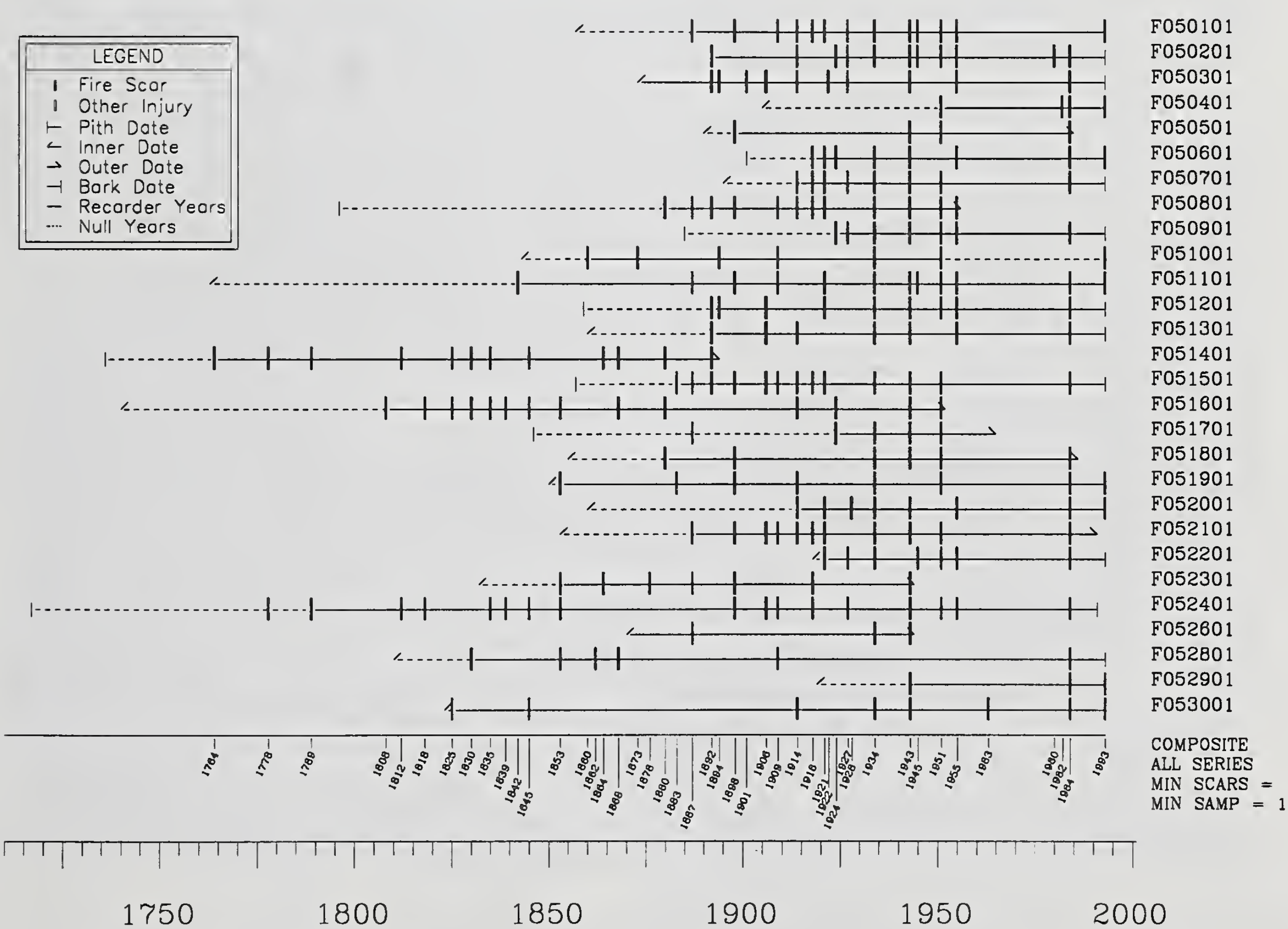

FIGURE 1. **Composite master fire chart for the Ejido Topia study site, Durango, Mexico. Frequent fires were interrupted at this site after 1955. Following a 29-year fire-free period, widespread fires returned in 1984 and 1993.**

MFI of 6.5 years (min. 3 years, max. 15 years, WMPI 6.4 years) for fires scarring at least 25% of recorder trees. However, a 29-year gap occurred between 1955 and 1984. The 1984 fire covered the entire study site and scarred almost all the sample trees. Nine years later, in 1993, another large fire again swept across the study site. The 29-year gap from 1955 to 1984 appears to represent an initial disruption of the frequent fire regime.

Similar disruptions were seen at two other study sites, where frequent fires stopped after 1945 and 1951, respectively. At the final site, in contrast, an uninterrupted frequent fire regime has continued up the present (Figure 2). Taken together, these four fire histories show that fire exclusion began in some parts of this region in the mid-twentieth century, but a great deal of variability is also seen in the record. One study site has continued to burn to the present, and the Ejido Topia fire history shows a three-decade fire exclusion period followed by the return of large fires.

Ecosystem structures

To understand the ecological role of fire we must ask: what is the impact on ecosystem structures of frequent fire regimes, their disruption, and the return of fire following an exclusion period? Specifically, what forest densities, tree sizes, and dead biomass loadings do we find in the presence of frequent low-intensity fires? How do these structures change when fire is excluded? And what happens when fire returns to the altered ecosystem?

We contrasted two paired unharvested pine-oak forest areas each 70 ha (173 acres) in size, which were closely matched in site characteristics (elevation, slope, aspect, slope position, soil parent material, and species composition). Both sites had frequent fires with mean fire return intervals around 4 years in the past. However, fire was excluded from the Arroyo Verde site since 1945, while fires continued regularly at the Arroyo Laureles site up to the present.

Figure 2. Unharvested pine-oak forest which has burned with frequent, low-intensity fires up to the present at the Arroyo Laureles study site, Durango, Mexico. The overstory of *Pinus durangensis* and *P. herrerai* consists of relatively large, widely-spaced trees which have been pruned by fire. Charcoal and small fire-scarred catfaces are evident on the lower boles. Reprinted from Fulé and Covington (1994) by permission of Blackwell Science, Inc.

Following a half-century of fire exclusion, ecosystem structures diverged substantially between the two sites (Table 1). The frequent fire site was dominated by large, well-spaced trees with an open forest floor (Figure 2), while the fire-excluded site became dense with small diameter trees. Age structure analysis showed that the two sites were fairly similar in density prior to fire exclusion, with most of the small trees becoming established after the final large fire on the exclusion site in 1945. The frequent fire site continues to have prolific regeneration potential, evidenced by high seedling germination, but these plants are repeatedly thinned by recurring fires, maintaining the open forest.

Forest fuels differed not only in volume but also in quality between the sites (Table 1). Rotten woody fuel loading was far higher at the fire-excluded site. These fuels dry rapidly and ignite easily, enhancing the flammability of the site. The live fuel structure at the fire-excluded site, composed of densely grouped trees in a variety of heights ranging from the forest floor to the dominant canopy, is also primed to carry fire rapidly into the overstory crowns and support high-intensity burning. In contrast, the trees at the frequent-fire site have been pruned of their lower branches by repeated fires, and few small trees exist to serve as a fuel ladder to the canopy.

The accumulation of forest floor fuels, combined with their increased flammability and the fuel ladder created by tightly-clumped living trees combine to support a shift to an infrequent, high-intensity fire regime at the fire-excluded site. Such a shift could have serious negative consequences. In economic terms, the mortality of 70 ha of mature forest would represent a major loss to the ejido members. In ecological terms, such a fire would represent a disturbance of unprecedented intensity. The site holds many old-growth trees over 100 or 200 years old, which have survived dozens of low-intensity wildfires. A crownfire would kill this genetic resource, heat the forest floor and expose soil to erosion, and open a large gap that may require many years for reforestation. While any individual 70 ha parcel in the Sierra Madre does not represent a great loss, the analogous experience of fire exclusion in the US shows that extensive contiguous areas of fuel can accumulate to support massive destructive crownfires (Covington et al. 1994, Everett et al. 1994).

An example from a fire in a similar forest—a ponderosa pine forest of northern Arizona—illustrates the importance of forest structure in influencing crownfire behavior. The Trick fire of July, 1993, burned for several days about 30 km (18 miles) northwest of Flagstaff, Arizona. In dry, windy weather, the high-intensity crownfire passed across an experimental thinning demonstration area (Figure 3). All trees in a densely-spaced stand were killed, even without the presence of a fuel ladder of smaller trees. In contrast, in a stand with a more open spacing, the fire caused only an incomplete surface burn. The key difference in stand density controlling fire behavior is striking because the two stands were adjacent and were burned over at the same time.

Finally, another of the Durango study sites presents an example of a disrupted fire regime followed by the return of fire. The Ejido Topia fire history showed a gap of 29 years following the last widespread fire in 1955 (Figure 1). The 1984 fire, burning in the accumulated fuel of three decades, appears to have had more intense behavior and caused more overstory mortality than the frequent fires which burned through the site every 4 to 6 years prior to 1955. Numerous overstory trees were killed and their replacements have tended to change the composition of the forest: old-growth pines and oaks were often replaced by densely-sprouting oak and alder species, forming a thick, shrubby growth. A reburn nine years later, in 1993, then burned through much of this shrub layer, often leaving the snags from the 1984 fire surrounded by dead alder and oak sprouts. In areas where pines germinated following the 1984 fire, they were often killed by the 1993 reburn. These fire effects were not uniform across the study site; many areas survived the 1984 and 1993 fires without much apparent old-growth mortality. However, the fire behavior and tree mortality appeared to be most intense in the most vulnerable parts of the system: dry, exposed slopes with thin, rocky soils where trees generally face challenges in

Table 1. Comparison of forest structure at two paired, unharvested pine-oak study sites in Durango, Mexico. Standard error of the mean is shown in parentheses. See Fulé and Covington (1994) for additional information.

Study site	Arroyo Verde		Arroyo Laureles	
Number of plots	30	30		
Fire regime	Fire exclusion after 1945, frequent fire (MFI = 3.8 years) from 1801-1945		Frequent fire (MFI = 4.7 years) from 1879-1992	
Density (trees/ha)				
Pine	1500	(235)	275	(43)
Oak	981	(107)	287	(47)
Other	253	(38)	85	(29)
Total	2730	(264)	647	(68)
Forest floor woody fuels (metric ton/ha)				
0-0.6 cm diameter	0.13	(0.02)	0.15	(0.02)
0.6-2.5 cm	0.56	(0.10)	0.54	(0.09)
2.5-7.6 cm	2.93	(0.53)	2.98	(0.78)
>7.6 cm sound	0.53	(0.27)	4.33	(1.50)
>7.6 cm rotten	11.65	(2.31)	2.56	(1.12)
Forest floor depth (cm)				
Litter	2.08	(0.15)	2.23	(0.16)
Duff	2.15	(0.18)	1.06	(0.09)

FIGURE 3. The Trick fire passed through adjacent *Pinus ponderosa* stands thinned to different density levels in the Coconino National Forest, Arizona, USA, in July, 1993. Trees in the denser stand (left) were completely killed by crown fire, while the more open stand (right) experienced an incomplete surface burn.

becoming established and where many of the oldest trees on the site are found. On some of these areas, the 1984 fire following 3 decades of fire exclusion may have caused a long-term deforestation.

To summarize, fire disturbance in the central Sierra Madre Occidental appears to be a keystone ecological process (*sensu* Holling 1992) regulating forest density, dead biomass, and regeneration dynamics. When frequent fire regimes are disrupted, forest density, fuel loading, and the vertical fuel continuity increase. The return of fire into a fire-excluded community can have effects quite different from those of the pre-disruption, low-intensity fires. Once subject to extended fire exclusion, the structural changes in the forest ecosystem can support high-intensity fire behavior, including crownfire and substantial overstory mortality. In effect, fire exclusion appears to lead to a switch from a regime of frequent, low-intensity fires to one of infrequent, high-intensity fires. If these changes are carried across broad landscapes, as has occurred in much of the western US, then the mountains of northern Mexico may become subject to unprecedented and unforeseen changes in ecological communities, biological diversity, and long-term sustainability of forest ecosystems.

IMPLICATIONS

What implications does the role of fire in the Sierra Madre have for the conservation and management of ecosystems both in northern Mexico and throughout the range of related fire-adapted pine forests in North America?

Within Mexico, the ecological role of fire should be considered as a key factor in forest management planning. Ironically, it is as a consequence of the traditional relaxed fire control policies permitting frequent fires to maintain open forests, that large conflagrations have generally not occurred (SARH 1994) and that fire management has not been considered highly important in forest management. However, modern Mexican attitudes to fire are changing. Landowners have a new realization of the value of timber, made possible by increasing investment in roads and machinery, and the public is exposed

to anti-fire advertising campaigns and a growing awareness of the environmental threat of deforestation. In many localities, people are proud of recent firefighting successes. Although the resources for full-scale wildland firefighting are generally still minimal (González-Caban and Sandberg 1989, SARH 1994), fire exclusion is increasing across the Sierra Madre. By seeking to more fully understand the ecological role of fire and to incorporate fire into forest management, Mexican managers may be able to apply fire in prescription as a tool to prevent or minimize the ecological problems that have been extensively described in this conference.

Second, to understand and adequately evaluate the role of fire implies a thorough scientific understanding of ecological structures and processes. Unharvested sites still under frequent fire regimes are essential for study, but these sites are disappearing rapidly in northern Mexico. Sampling should cover a range of sites, from the high-elevation central Sierra Madre to the lower foothill forests and woodlands. Experimental approaches, including thinning, fuel treatments, and prescribed burning, should be undertaken in addition to descriptive studies.

A third implication of fire's ecological role is that true ecological conservation includes the conservation of disturbance regimes. In the US we see many examples of "preserved" forests, carefully protected from cutting, livestock grazing, or the encroachment of exotic species, but which nonetheless have undergone drastic and deleterious changes due to fire exclusion. An example is the North Rim of Grand Canyon National Park, where fire exclusion has contributed to increased forest density, meadow encroachment by trees, and invasion of pine forest by fire-intolerant firs. Since ecosystems are dynamic, we cannot hope to preserve one part of the system, such as the vegetation structure, without also taking into account the role of ecological processes, such as fire.

Finally, managers and scientists concerned with the health and sustainability of fire-adapted forests in the US and Canada can learn from closely-related intact Mexican ecosystems. Data from Mexican frequent-fire sites can broaden our understanding of fire's dynamics and variability. Comparing sites with varying periods of time since fire exclusion can clarify the direction and speed of ecological change following fire regime disruption. In trying to restore the balance of degraded North American ecosystems, forests of northern Mexico can present a benchmark for comparison and a testing ground for ideas about ecological restoration. Aldo Leopold concluded his 1937 description of the Sierra Madre Occidental by calling for us to collaborate in this way, saying: "...the Sierra Madre offers us the chance to describe, and define, in actual ecological measurements, the lineaments and physiology of an unspoiled mountain landscape. What is the mechanism of a natural forest? A natural watershed? A natural deer herd? A natural turkey range? On our side of the line we have few or no natural samples left to measure. I can see here the opportunity for a great international research enterprise which will explain our own history and enlighten the joint task of profiting by its mistakes."

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To take up the torch: The hidden risks of fire exclusion; how and why we must turn the flame to its rightful place in the West

Bruce Babbitt[1]

Just north of Interstate 70 in the Colorado Rockies rise the now charred slopes of Storm King Mountain. There, at 4 p.m. on July 6, 1994, a wall of flame 300 feet high swept uphill, snuffing out the lives of 14 fire fighters who stood in its path.

The mountain seems an unlikely place for such a conflagration. Its unburned flanks are covered only with scraggly patches of pinon-juniper and Gambel oak, rooted in open expanses of red rock and dirt. Hardly a forest, and legally it isn't; when they drew the boundaries of the nearby White River National Forest in 1900, foresters didn't bother to include these scrubby slopes.

Yet shrublands like these are what really ignited the great fire summer of 1994. Granted, it was an unusually hot summer of clear skies punctuated by dry lightning, but in these desert lands, drought is no stranger. It was other forces, human forces, that have transformed these arid lands into a landscape of fiery destruction.

I witnessed those forces firsthand. One week before the fatal blowup at Storm King, I had joined a fire crew on the Bunniger Canyon Fire, burning just north of Grand Junction. A helicopter set us on a razorback ridge to dig a fireline across a slope where the fire had burned down from the mesa top. We anchored the line on a dusty slope where junipers grew widely spaced apart, yet as we moved into the drainages, thick clusters of Gambel oak and Utah serviceberry surrounded us everywhere. Time after time, the growth forced us to stop, put down our shovels and Pulaskis and wait for saw crews to hack through the thickets.

And it wasn't just that slope. Across the entire western landscape, from Mexico to the Canadian border, scrub trees are taking over: junipers advance across lowland plains; doghair ponderosa fill gaps in the highland forests; spruce and fir crowd out aspen groves.

Why are our western forests and rangelands changing so dramatically? Because we have systematically removed the natural flame. Just as we wiped out the wolf that preyed on weak, sick, and overpopulated herds, we have eliminated the frequent, light-burning fire cycles that used to thin the forests of young trees, kill off the spreading juniper seedlings and hold brush in check.

The naturalist Aldo Leopold, then an Arizona forester and fire fighter himself, first recognized the extent of our impact in 1924. He observed a sharp contrast in the age grouping of Arizona junipers, ancient, fire-scarred trees that stood in a matrix of very young trees all less than forty years old, with no age groups in between. Leopold surmised that beginning in the 1880's something had intervened to keep fires from spreading after ignition:

"Previous to the settlement of the country, fires started by lightning and Indians kept the brush thin, kept the junipers and other woodland species decimated, and gave grass the upper hand with respect to the possession of the soil ... then came the settlers with their great herds of livestock. These ranges had never been grazed and they grazed them to death, thus removing the grass and automatically checking the possibilities of widespread fires."

Even as Leopold wrote those words in 1924, the U.S. Forest Service had begun a campaign to exclude fire across the continent. Sparked by the fires of 1910 in the northern Rockies, and prodded by Washington, the Forest Service took up fire suppression with a vengeance. Smokey Bear urged prevention at all costs. Airplanes that dropped paratroopers and bombs during World War II now spawned smoke jumpers, fire retardants, and chemicals, all with the singular target of pinching out every fire by 10 a.m. the next morning. It was an effective campaign. So effective, in fact, that even today it often mutes any suggestion that in some cases fire **improved** the health of ranges and the forests, and that there is a risk of excluding fire as well.

Paradoxically, as fire exclusion escalates, wildfires fight back with increasing ferocity. In the absence of fire, ground fuel accumulates and crowded forests become more susceptible to disease and insect damage. So when lightning inevitably strikes, the odds are much higher that it will flare up faster, burn hotter and higher, crown into the big trees and decimate entire forests in what professionals call a "stand replacing fire." These intense, densely-fueled wildfires are also increasingly expensive, and unpredictable, to fight.

[1] *Secretary of the Interior, Washington, D.C.*

The only way to break this vicious cycle is to put controlled fire back onto the land. We must apply the torch to recreate the prehistoric cycles of light burning where ground fires moved swiftly across the land, consuming brush and accumulated ground fuel, pruning out thickets and maintaining healthy stands of forests.

In some parts of the country, land managers do regularly use these controlled burns, or "prescribed fires," to boost both the local ecology and economy. In Southern forests they burn back the hardwood understory to stimulate germination and growth of pines. In the Flint Hills of Kansas, ranchers burn back the tall grass prairie each winter to promote a vigorous new spring growth. And rather than letting Santa Ana winds blow wildfire out of control, southern Californians have begun the regular, controlled burning of the chaparral lands that surround them.

Another advantage of prescribed fire is timing. Wildfires typically ignite at the worst time - during the dry "fire season," when they can break out of control and when manpower and equipment are stretched dangerously thin. By contrast, prescribed fire allows us to choose weather, temperature, and season for burning, often in the spring or fall when the air is cool and moist enough to keep fire within limits. Also, land managers have time to plan and construct adequate fire breaks, or to reduce the fuel load by hand thinning around valuable sites and trees.

Yet despite mounting evidence of the benefits, prescribed fire is still not widely used in the West. From 1984 to 1993, on 270 million acres of Bureau of Land Management (BLM) lands, wild and prescribed fire burned an average of 960,000 acres per year. At that rate, a given acre of BLM land would burn once every 284 years; an acre of Forest Service land would burn once every 230 years.

The calculation is rough; some desert lands would not burn under any conditions, while old growth forests of the Pacific Northwest burned historically perhaps every thousand years. However, the vast majority of western public lands, including rangelands, chaparral, and ponderosa forests burned historically every 10 to 50 years. Prescribed burning should approach that historic level.

So why have we been slow to take up the torch? For many years the Smokey Bear-educated public saw only the risks of fire, not the benefits. And indeed, when a million acres of Yellowstone burned in 1988, the initial public response was highly negative. But when visitors saw the miraculous cycle of renewal, purple fields of blooming fireweed and slopes greening with lodgepole seedlings, attitudes began to change. Yellowstone taught us, in a most spectacular and instructive setting, that fires are a natural and necessary part of ecological succession. Last October, *American Forests* confirmed this shift in a careful poll: in California, 55 percent favor controlled burning, as do two thirds of respondents in the Inland West.

Yet, even as overall public attitudes shift toward acceptance, the site specific, anywhere-but-in-my-backyard, objections are hard to overcome. In Arizona, for example, the prevailing winds sometimes shift during prescribed burns on the Mogollon Rim, pouring smoke downhill into the inversion basin over Phoenix. And that triggers angry responses from the public. But the fact remains: we either pay now with some inconvenience, or we will surely pay a higher price later with larger, smokier, uncontrollable wildfires. Prescribed fire plans will require careful coordination with the Environmental Protection Agency and state air quality regulators.

Similarly, the liability issues are quite real, no prescribed fire is ever 100 percent escape proof and property damage can and does occur. That fear of liability can paralyze prescribed fire managers at any level. Yet the alternative of allowing fuel to build up to feed the inevitable big wildfires is even worse, as hillside residents in southern California can readily testify. Our challenge is to assess those risks, and work out cooperative protection agreements with participating landowners.

Apart from these obstacles, prescribed fire is not being used with optimum effect because we in the land management business have not been its forceful advocate. If we gave it just a fraction of the time and energy that our predecessors put into the fire exclusion campaigns, prescribed fire would soon take its rightful place on the land management agenda.

To bring prescribed fire up to its full potential for restoring western forests and rangelands will require concerted action at both the federal and state level. A first essential step is for the federal agencies to elevate prescribed fire to full status in the federal land use planning process. Both the Forest Service and the BLM are required by law to produce and regularly update land management plans at the forest and district level. Yet even a casual sampling of current plans reveals how little attention is paid to prescribed fire; most plans do not even discuss the concept, much less undertake serious analysis. Even environmental organizations, usually so quick to prod federal agencies with lawsuits challenging the adequacy of the planning process, seem to have entirely overlooked the use of fire as a management alternative important enough to require discussion in virtually all land use plans.

Plans for the use of prescribed fire must include the states and their political subdivisions, for it makes little ecological or economic sense to confine prescribed fire within federal fences, when the benefits could be extended to all landowners, state, federal, and private. Fortunately, there is a good precedent right at hand. In

1911, a time when fire suppression efforts often failed for lack of coordination, Congress enacted the Weeks Act. The Act, and successive legislation, provided matching grants to those states willing to adopt comprehensive fire suppression plans acceptable to both the state and the Forest Service.

The time is now at hand to expand this proven federal/state partnership beyond fire exclusion to the broader objective of introducing fire onto the landscape as a routine management tool. Congress could extend existing Federal cooperative grants to require that states, to be eligible for existing revenue sharing, must produce prescribed fire plans acceptable to major federal and state land agencies.

Arguably, we do not even need legislation, for the 1978 Weeks Act amendments expressly authorize the Secretary of Agriculture to provide assistance to the states to plan and organize programs of "prescribed burning." In the 17 years since those words were written into law, individual agencies have made sporadic progress, yet the development of true statewide multiagency plans remains to be achieved.

In the end, however, plans are just so much paper without the leadership and money to put them into effect. Comprehensive prescribed fire plans will require additional funds. The logical source of funding is revenue produced by the public lands. Just as rents from a building are the source of funds for the maintenance and upkeep of the asset, so the receipts from the products of the land, like timber sales and grazing fees, should be earmarked for upkeep of the land through the use of fire to invigorate and renew the range and forest resources.

Ideally, funding would consist of a single Federal appropriation from public land revenues, to be apportioned among federal agencies in proportion to their land base within a given state. A unitary appropriation should also provide matching funds for states to carry out prescribed fire on state land. And there is no reason why similar matching incentives should not be extended to the owners of non-commercial private land, a concept that is already used by the Department of Agriculture in other forestry programs.

A comprehensive movement that puts prescribed fire back on to the landscape, that increases the health and productivity of the land, and that reduces the risks and destruction of wildfires that do occur, would be a lasting memorial to the brave firefighters who lost their lives during the summer of 1994.

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